An Assessment of the Environmental Performance of Seafood and the Implications for Food Systems

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“The health of humans cannot be isolated from the health of ecosystems”

FAO 2012 - Sustainable diets and biodiversity

DECLARATIONS

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ABSTRACT

Food security is underpinned by the sustainability of the global food system. The production and supply of food is responsible for global environmental impacts and a transition to a sustainable food system is required. Seafood plays a key role in the food system, as a vital source of protein and essential nutrients, yet fisheries and aquaculture are often left out of discussions on food and nutrition security. The broader impacts of seafood as part of the food system are rarely considered through current sustainability assessments. The result is that the environmental impacts generated from the supply and consumption of seafood products are not well understood. Life cycle assessment (LCA) is a practical and credible tool for assessing the environmental footprints of food products and production methods. LCAs of seafood have increased substantially over the past decade, encompassing a range of species, production methods, and geographic locations. Nevertheless, the field lags behind LCA of terrestrial systems in terms of both empirical coverage and methods.

This thesis expands the literature on seafood LCAs, using Australian seafood case studies. The aim is to improve the understanding of the environmental impacts of seafood production and consumption as a part of the broader food system, and to identify opportunities to advance seafood sustainability concepts and practice. The thesis consists of five papers which are linked through the LCA framework, and have a strong emphasis on seafood as part of a sustainable food system, and on the nexus between LCA and contemporary fisheries management principles and practices. In the first paper LCA is used to measure the environmental performance of the white banana prawn (*Fenneropenaeus merguiensis*) from the Australian Northern Prawn Fishery across its supply chain. Management of this fishery has been promoted as a sustainable model for other countries to emulate, although broader environmental impacts, such as those relating to energy and water use or greenhouse gas emissions are not currently monitored. Fishing operations were the main source of impacts, while processing and storage were key contributors to ecotoxicity, and transport made a negligible contribution to any impact category. This research highlighted the scope to develop the application of LCA in wild capture fisheries in terms of complementing existing fisheries management, and through the development of fishery-specific indicators to improve the efficacy of seafood LCAs.

There is a need for information to better understand the relationship between seafood LCAs and fisheries management. In the second paper LCA was used to measure the environmental footprint of the supply of Tasmanian southern rock lobster, *Jasus edwardsii* (TSRL) under different management

scenarios. The environmental footprint of the TSRL in the scenarios modelled was responsive to marine resource management decisions made inside and outside the fishery. Targeting maximum economic yield rather than maximum sustainable yield decreased the carbon footprint by 80%. Limiting access to the fishery by increasing the coverage of marine protected areas increased the fishery's carbon footprint by 23%. Better understanding of the environmental impacts resulting from management changes will be vital in a future of increased carbon emission reporting and regulation. The application of LCA to marine resource decision-making can help ensure decisions are not made in isolation of broader environmental impacts.

The lobster LCA also highlighted that the international airfreight of live lobsters was the major contributor to global warming potential (GWP) and cumulative energy demand (CED). The distance between where food is produced and consumed is increasing, and is often taken as evidence of an unsustainable global food system. Seafood is a highly traded commodity and in the third paper LCA was used to examine the impacts of the movement of products beyond the fishery or farm to better understand the environmental impact of seafood as part of the broader food system. The carbon footprints of the production and distribution of select seafood products that are consumed in Australia were compared to determine differences in the sustainability of imports and their domestically produced counterparts. The distance food is transported was not found to be the main determinant of carbon emissions. Despite the increased distance between production and consumption, carbon footprints of meals from imported seafood can be similar to meals consisting of domestically produced seafood, and sometimes lower, depending on the seafood consumed. In combining LCA with existing seafood sustainability criteria the trade-offs between sustainability targets become more apparent.

While the addition of carbon footprinting to current seafood sustainability assessments broadens their perspective, the carbon footprint should not be taken as a complete measure of product sustainability in LCA. A combination of relevant LCA indicators can provide a more holistic assessment of overall seafood sustainability. However, while LCA is a well-developed and useful tool for assessing carbon emissions and other biophysical impacts, it is lacking in the assessment of fishery specific impacts. Several new methods have been proposed in recent years to account for fishing impacts on ecosystems and biotic resources through LCA. In the fourth paper, a new method is proposed to complement these advances through accounting for impacts of fishing on biodiversity as a measure of naturalness of the seafloor and pelagic habitats. The method has been adapted from its original terrestrial application and incorporates elements of existing seafood certification schemes and marine ecological risk assessment. It is a contribution to the much needed improved application of LCA to wild-capture fisheries.

In the final paper, a literature review is conducted to examine how the environmental performance of seafood is integrated and interpreted within the rapidly growing body of literature on sustainable diets. Seafood is examined in terms of its reported comparative performance to agricultural products and in its role in a sustainable food system. Seafood diets typically have lower carbon footprints than meat diets and higher footprints than vegetarian diets. However, many studies do not adequately address seafood and thereby overlook the opportunities and limitations of including seafood as part of a sustainable diet.

LCA is used in this thesis to assess the environmental impacts of seafood and to expand the concept of sustainability of seafood, however, fisheries, aquaculture and their supply chains remain often neglected, yet critically important, parts of healthy and sustainable food systems. This research brings seafood a step closer to adequate representation in food systems research, although further development of seafood LCAs and better methods for comparing seafood and agricultural production systems are required. More of this type of research is therefore needed to ensure that seafood is included in future food and nutrition security discourse.

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This thesis represents my foray into fisheries, how they are managed, the challenges they face and their contribution to the global food system. I feel incredibly lucky to have had the opportunity to explore this dynamic and important area at a time when it seemed to be coming of age and was fortunate to be involved in a number of national and international projects in this research field.

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CHAPTER 1: GENERAL INTRODUCTION

1.1 Global food system and food security

The food system is a major source of global environmental impacts, driven by food consumption and production trends (Tilman and Clark 2014; Godfray et al. 2010). The agriculture sector is the largest consumer of water globally (Strzepek and Boehlert 2010; Gleick et al. 2009) and croplands and pastures occupy around 40% of the land surface (Foley et al. 2005). The production of food is reliant on fossil fuels as the main source of energy (Woods et al. 2010) and approximately one-quarter of global greenhouse gas emissions (GHGe) result from land clearing, crop production, and fertilisation (Burney et al. 2010). Marine, freshwater, and terrestrial ecosystems have been damaged by fertiliser and pesticide use (Matson et al. 1997; Vitousek et al. 1997) and ongoing habitat fragmentation threatens biodiversity (Butchart et al. 2010). As the population increases in number and wealth, growing demand for food will place further pressure on land and other natural resources (van der Werf et al. 2014). The provision of food for future generations will be challenged by the loss of ecosystem services (Millennium Ecosystem Assessment 2005) and the effects of climate change, which are expected to impact both the production and supply of food and potentially place more people at risk of food insecurity (Schmidhuber and Tubiello 2007).

Shifting to a more sustainable food system is required to achieve global food security without depleting natural resources and further degrading natural ecosystem functions (Berry et al. 2015). The need for a more sustainable food system is widely recognised (Garnett 2014; Allen and Prosperi 2016; Soussana 2014; van der Werf et al. 2014; Slade 2013) with the environment seen as the missing, yet underpinning, fifth pillar of food security (UNEP 2012) (Figure 1.1).

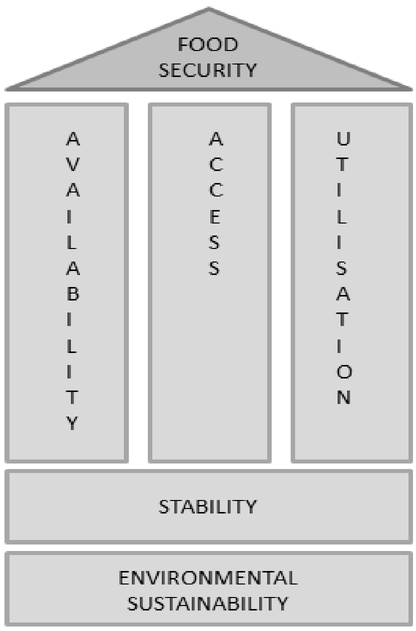

Figure 1.1 Five pillars of food security (adapted from UNEP 2012)

Food systems are social–ecological systems with multiple interactions between human and natural components, and can be examined from many different theoretical viewpoints. The framework of this thesis is based on the environmental dimensions of the more complex system, although there are other equally valid frameworks such as those of vulnerability and resilience (Prosperi et al. 2016; Tendall et al. 2015) and human equity (Agyeman 2008). While the focus of this thesis is on environmental impacts, it is important to note that environmental justice and equity are also vital elements of sustainability (Agyeman 2008; Lam and Pitcher 2012; Agyeman et al. 2002) and evaluating sustainability from purely an ecological perspective restricts its scope and definition (Loring 2013; Richmond 2013). Here, the concept of a sustainable food system is based on the idea that all activities related to food, including producing, processing, transporting, storing, marketing and consuming, are interconnected and interactive (UNEP 2012). A range of methods have been developed to better measure and address the environmental impacts from these activities, including risk assessment, environmental performance evaluation, environmental auditing, environmental impact assessment, and life cycle assessment (LCA) (ISO 2006a). Of these methods, LCA has emerged as a useful and practical tool capable of quantifying the interconnected environmental consequences of food systems. LCA is an International Organization for Standardization (ISO)-standardised accounting framework used to quantify the environmental impacts associated with the energetic and material intensity of products or processes across their supply chains (Pelletier and Tyedmers 2008). LCA has evolved from its original application to manufacturing processes in industrial production systems (Harris and Narayanaswamy 2009; Horne et al. 2009) and is now used within business, academia and policy to assess the environmental footprints of food products and production methods (de Vries and de Boer 2010; Henriksson et al. 2012a; Harris and Narayanaswamy 2009; Nijdam et al. 2012; Roy et al. 2009), identify options for reducing the impacts of the food system (UNEP and SETAC 2009), and identify trade-offs between environmental sustainability and health to achieve food security outcomes (Garnett 2014).

1.2 Seafood as part of the global food system

The relationships between environmental change, food, and health are highly complex (McMichael et al. 2007) and global strategies to address these relationships are predominantly focused on terrestrial systems. Limited attention has been given to seafood[1] as an important element of food security and nutrition strategies at national and global levels (HLPE 2014b; Olson et al. 2014; Béné et al. 2015; Lang and Heasman 2009; Smith et al. 2010). Yet seafood is vitally important for global food security, both in terms of providing a direct source of food, and indirectly as a means of ensuring access to food through its contributions to incomes and economic growth (Béné et al. 2015). Seafood accounts for 16.7% of the global population's intake of animal protein and 6.5% of all protein consumed (FAO 2014b). Seafood is also an important source of essential micronutrients

[1] The word seafood is used to describe fish and invertebrates from wild-capture fisheries and aquaculture

(HLPE 2014b) and the very long chain polyunsaturated fatty acids, eicosapentaenoic acid (EPA) and docosahexaenoic acid (DHA) (Lund 2013). Regular consumption of seafood has been linked with lower risks of a range of health conditions, including cardiovascular disease, stroke and dementia (Weichselbaum et al. 2013; Nestel et al. 2015; Larsson and Orsini 2011).

The fisheries and aquaculture sectors together provide a source of direct income for over 58 million people (FAO 2014b), and indirectly through secondary activities for 540 million people, or 8% of the world population (FAO 2011). Seafood is one of the most highly traded commodities and seafood trade represents a significant source of foreign currency earnings for developing countries, more than the combined total of other agricultural commodities such as rice, coffee and tea (FAO 2014b). Actual consumption of seafood is low in some developing countries, while developed countries typically have high consumption levels (Smith et al. 2010) and account for over 70% of total fisheries imports by value as domestic production cannot meet demand (FAO 2014a). The mismatch between areas of fisheries and aquaculture production with areas of demand contributes to the very high levels of trade in fish and fish products (Watson et al. 2015b).

Globally the supply of seafood is increasing at an average annual rate of 3.2%, which is faster than population growth (FAO 2014b). Aquaculture has been the world's fastest growing food production sector for more than four decades (Tveteras et al. 2012) and provides an increasing share of total seafood supply, while global capture fishery production has stabilised at around 80 million tonnes. Aquaculture contributes significantly to global *per capita* animal protein consumption, with production now surpassing beef production (Jennings et al. 2016). However, unlike wild-capture production, many aquaculture systems are dependent on other production industries for feeds (Troell et al. 2014). The reliance of finfish aquaculture on wild fish oil and meal for feed, coupled with growing demand for seafood and the historical overexploitation of fisheries, has led to concerns over the sustainability of global fisheries (Garcia and Grainger 2005; Worm et al. 2009).

Although seafood consumption drives pressure on fish stocks, the focus of fisheries policy, management and research has been on fish stocks as renewable, but exhaustible, natural resources, rather than as part of the food system. Food production is rarely mentioned as a guiding goal or objective in fishery management (Olson et al. 2014). The sustainability of seafood has been associated with the condition of the resource being exploited. Fisheries are generally considered to be sustainable when stocks have an abundance at or above the level that can produce the maximum sustainable yield (MSY) (FAO 2014b). Stocks fished to a level below MSY are considered to be overfished, and generally thought to be unsustainable, although there remains great differences in both perception and definition of the concept of sustainable fisheries (Hilborn et al. 2015).

The habitats and ecosystems that sustain fish populations can also be degraded through fishing, or altered over time so that sustainable fisheries may operate in non-natural systems (Pinnegar and Engelhard 2007). Ecosystem-based fisheries management (EBFM) is as an approach to sustain healthy marine ecosystems, and the fisheries they support, by addressing some of the unintended consequences of fishing, such as habitat destruction, incidental mortality of non-target species, and changes in the structure and function of ecosystems (Pikitch et al. 2004). Ecosystem impacts of fishing have also been considered by environmental non-government organisations and incorporated in to seafood certification and ecolabelling processes of organisations such as the Marine Stewardship Council (MSC) and consumer awareness programs such as the Monterey Bay Aquarium's 'Seafood Watch' program (Pelletier and Tyedmers 2008).

1.3 Seafood impacts measured through LCA

While assessment and management of the direct biological impacts of fishing has improved, the indirect and off-site effects of fishing activities have been largely ignored until recently (Avadí and Fréon 2013). The material and energetic demands of industrial fisheries can result in considerable impacts (Pelletier et al. 2007), ranging from the provision of fishing gear (Ziegler et al. 2003) and bait (Driscoll and Tyedmers 2010), to the use of fossil fuels in vessels (Parker and Tyedmers 2014; Parker et al. 2014b; Driscoll and Tyedmers 2010; Freon et al. 2014; Ziegler et al. 2011; Ziegler and Hansson 2003; Iribarren et al. 2011), and the transportation and processing (such as fileting and canning) of landings (Andersen 2002; Hospido et al. 2006; Ziegler et al. 2013; Almeida et al. 2015; Vazquez-Rowe et al. 2014). In particular, the fuel that a fishery uses and the resultant greenhouse gas emissions (GHGe) are an important aspect of seafood sustainability (Driscoll and Tyedmers 2010). Fisheries account for about 1.2% of global oil consumption (Tyedmers et al. 2005) and the energy performance of fishing fleets has reportedly declined over time (Mitchell and Cleveland 1993; Tyedmers 2001, 2004) as a result of vessels needing to search longer and to fish deeper in offshore waters as coastal stocks decline (Morato et al. 2006; Watson et al. 2015a). Improvements in fuel use efficiency have been noted more recently and further improvement is reliant on the ability of fisheries management to rebuild stocks and reduce over-capacity (Parker and Tyedmers 2014).

Life cycle considerations are generally not addressed through fisheries management and sustainable seafood programs (Pelletier and Tyedmers 2008), however, research on the life cycle impacts of seafood has increased substantially in recent years. LCA has been applied to aquaculture and fisheries since the early 2000s, and while the number and scope of case studies are growing (Avadí and Fréon 2013), many are focussed on a small number of key species from Europe or on those species destined for European markets (Parker 2012). Case studies from the Southern Hemisphere and developing countries are less common, despite the significant amount of seafood sourced from these areas. Seafood LCA studies have tended to focus on GHGe, with few including a broader suite of impact categories as suggested by the ISO standards (ISO 2006b). In particular, the direct effects of fishing on marine ecosystems are not commonly considered within the context of an integrated life cycle approach (Avadí and Fréon 2013). Due to the industrial focus of early LCAs a lower level of maturity in the LCA methods and techniques is seen across assessment of food systems in comparison to industrial systems. For example, less data are available relating to biodiversity change than energy (Horne et al. 2009). The ability of LCA to cover the wide range of environmental impacts potentially linked to fishing is dependent on further development of methods (Vázquez-Rowe et al. 2012a).

1.4 Thesis goals

In this thesis I use Australian wild-capture seafood case studies to measure the broader environmental impacts of seafood as part of the food system. The aim of the research is to improve the understanding of the environmental impacts of seafood production and consumption and to identify opportunities to advance seafood sustainability concepts and practice. This aim is addressed through empirical examination of the environmental performance of fisheries and their associated supply chains, and through development of the LCA method for application to wild-capture fishery products. The thesis consists of five papers which are linked through the LCA framework, and have a strong emphasis on seafood as part of a sustainable food system, and on the nexus between LCA and contemporary fisheries management principles and practices.

This thesis begins with an LCA of an Australian wild-capture fishery and progresses through specific life cycle aspects of seafood production and consumption. In Chapter 2, I explore the concept of ‘sustainability’ in relation to seafood and employ LCA to examine a broad range of environmental impacts in the MSC certified Northern Prawn Fishery. I identify opportunities for LCA to complement existing fisheries management and to broaden the scope of current seafood sustainability assessments. The relationship between fisheries management and LCA is examined further in Chapter 3 through quantification of the environmental impacts of the supply of Tasmanian southern rock lobster, *Jasus edwardsii* (TSRL) under different management scenarios. While management strongly influenced the environmental footprint of fishing, the substantial contribution of the export stage to the overall product footprint highlights the importance of impacts occurring as a result of the movement of seafood beyond the fishery, and beyond national borders. I examine the transport of seafood products in more detail in Chapter 4 to determine differences in carbon footprints between domestically produced and imported seafood. The chapter results confirm that production and transportation mode are more important considerations than distance , and that the use of food miles as a sustainability metric, in isolation from additional metrics, ignores other supply chain stages and environmental considerations, potentially overshadowing more relevant indicators that are important for balanced debate on food sustainability.

Quantifying carbon footprints of seafood during fishing, and beyond the fishery, provides a broader perspective of seafood sustainability, however, a holistic assessment of product environmental performance should account for both the direct and indirect environmental impacts. Assessments of direct biological impacts are not well integrated into LCA and there is a need for further methodological development. Chapter 5 builds on recent developments in fishery-specific LCA methods and on fishery management assessments, through development of a new method to assess the impacts of fishing on the naturalness of marine systems. The method further highlights the opportunity to combine LCA and fisheries management, as outlined in Chapters 2 and 3. Chapters 2-5 use LCA to assess aspects of seafood sustainability, thereby developing a picture of the environmental performance of selected seafoods. In Chapter 6 I examine how the environmental performance of seafood, determined primarily through LCA, is integrated and interpreted within the rapidly growing body of literature on sustainable diets. I highlight deficiencies in the manner in which seafood is dealt with in discourses about the role of seafood in future food systems and the opportunities and limitations of including seafood as part of a sustainable diet.

In this thesis I use LCA to measure a range of indicators of seafood sustainability beyond traditional stock-assessments. In particular I examine the efficacy of LCA to assess the environmental impacts of seafood, and how these assessments can contribute to traditional fisheries management and sustainable food systems. Each chapter considers aspects of sustainability relevant to contemporary fishery management and independent third-party assessments, as well as a broader range of life-cycle impacts. The overarching goal is to assess how broader food system impacts can be included in the measurement of sustainable seafood. This research is significant given the potential, yet often overlooked, positive role that seafood can play in shifting toward sustainable food systems to feed a growing population in the wake of a changing climate.

CHAPTER 2: LIFE CYCLE ASSESSMENT OF WILD CAPTURE PRAWNS: EXPANDING SUSTAINABILITY CONSIDERATIONS IN THE AUSTRALIAN NORTHERN PRAWN FISHERY

This chapter previously published as:

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2.1 Abstract

Prawns and shrimp are among the most popular seafood consumed globally and their production is responsible for a range of environmental impacts in wild capture fisheries and associated supply chains. Management of the Australian Northern Prawn Fishery has been promoted as a sustainable model for other countries to emulate, although broader envionmental impacts, such as those relating to energy and water use or greenhouse gas emmissions are not currently monitored under sustainability assessments. Life cycle assessment (LCA) is used to assess the environmental impacts of the white banana prawn (*Fenneropenaeus merguiensis*). Fishing operations were the main source of impacts for the supply chain examined, contributing 4.3 kg CO_2e kg^{-1} prawn or 63% of the overall global warming potential. This result was lower than emissions reported for other prawn species, including tiger prawns from the same fishery. Processing and storage were key contributors to ecotoxicity while transport made a negligible contribution to any impact category. Opportunities are presented for LCA to complement existing fisheries management, and broaden current seafood sustainability assessments, including the potential for emerging fishery-specific indicators to improve the efficacy of seafood LCAs.

2.2 Introduction

Prawns[2] are among the most popular seafood consumed globally and are one of the most important traded fishery products, accounting for 15% of the total value of internationally traded fish products (FAO 2010a). Globally, approximately 6.5 million tonnes are produced annually with around half of this production from wild capture fisheries and the rest from prawn farms (FAO 2012). Despite the

[2] 'Prawn' refers to both shrimp and prawn within Caridea and Dendrobranchiata.

recent growth of aquaculture production, wild capture prawns remain an important source of food and fisher livelihoods (Bondad-Reantaso et al. 2012). Prawn fishing and the use of bottom-trawls has been linked to a range of environmental impacts, the extent and reversibility of which vary with trawl type and location (Pitcher et al. 2009a; Brewer et al. 2006; de Groot 1984; Eayrs 2007), but it is the management of bycatch and discards that has dominated sustainability discussions around prawn fisheries for decades (Gillett 2008). Tropical prawn trawl fisheries accounted for over 27% of total estimated discards in global marine fisheries, or over 1.8 million tonnes per year (Kelleher 2005).

Negative consequences of trawling, such as the incidental mortality of non-target species, have notably improved over the last decade (He and Balzano 2011) with the emergence of ecosystem-based fisheries management (EBFM). However, the ability of EBFM to sustain healthy marine ecosystems and the fisheries they support (Zhou et al. 2010) will be continuously challenged by outside pressures, including climate change, which will have potentially detrimental consequences for some fisheries (IPCC 2014). Crustacean fisheries directly contribute to climate change through greenhouse gas emissions (GHGe) from the burning of fuel, and are characterised by the highest fuel use intensities in fisheries (Parker et al. 2014a; Parker and Tyedmers 2014). Prawn fisheries can also have very high energy use for the amount of food produced (Gillett 2008). Energy use, and the resulting emissions, are not typically included in sustainability assessments of prawn fisheries and their products, or in seafood more generally. Methods such as emergy accounting (Wilfart et al. 2013; Zhang et al. 2012) and Life Cyclce Assessment (LCA) have emerged to evaluate energy efficiency and carbon emissions along product supply chains, however the use of fossil fuel in fishing vessels has largely been excluded from the ecosystem approach (Pelletier et al. 2007).

GHGe and use of resources such as water are increasingly of interest in regard to food sustainability and security, as climate change alters food systems; and good quality water becomes scarcer in arid countries like Australia. The regulation of GHGe is expanding with 16 emissions trading schemes, covering 70 % of global emissions, expected to be in place by 2015 (ICAP 2014). The absence of these indicators from sustainability assessments undertaken by government, industry or certification groups such as the Marine Stewardship Council (MSC) means that current levels of impacts are not well understood. As a result, improvements through time have not been monitored as has occurred with other areas of improving fisheries practice such as bycatch reduction.

LCA has emerged as a standardised environmental management tool capable of analysing environmental burdens along the supply chain of products and processes (ISO 2006a, b). There has been a recent surge of LCAs in seafood systems, however, further development of the methodolgy is required in order to effectively cover the wide range of environmental impacts linked to fishing (Vázquez-Rowe et al. 2012a).

Australia's largest prawn fishery, the Northern Prawn Fishery (NPF), is considered sustainable under third party assessments, including those conducted by the Marine Stewardship Council (MSC 2012) and under the *Environment Protection and Biodiversity Conservation Act 1999* (FRDC 2012) of the Australian Commonwealth Government. These assessments cover a specific range of environmental criteria, however, they do not consider impacts relating to resource use, GHGe, or emissions of nutrients and toxins during fishing or along the supply chain. Neither do they address differences in these impacts between target species. LCA is used in this study to examine the impacts of the supply of 1 kg of banana prawn from the NPF. Opportunities for LCA to complement existing fisheries management are discussed as well as the potential for emerging fishery-specific indicators to improve the efficacy of seafood LCAs. This fishery was selected as a case study based on its importance to the Australian economy and because it was one of the first Australian fisheries demonstrating the ecological sustainability of its supporting ecosystem (Zhou and Griffiths 2008). This research adds an Australasian example to the growing body of seafood LCA literature and provides an example of how LCA can augment current concepts of seafood sustainability by broadening the scope of environmental considerations.

2.3 Methods

2.3.1 Northern Prawn Fishery case study

The NPF is the most valuable fishery managed by the Commonwealth Government of Australia with gross value of production of $94.8 million in 2010-11, accounting for over 4% of the total for Australian fisheries and aquaculture (Woodhams et al. 2012). The NPF is a multispecies fishery comprising 52 boats using otter-trawl gear in 2010/11. A total of 9,673 tonnes were landed in the same period, with white banana (*Fenneropenaeus merguiensis*) and tiger prawns (*Penaeus esculentus*; *P. semisulcatus*) accounting for 80% of the total annual catch (Woodhams et al. 2011). A number of byproduct species are also landed, including endeavour prawns (*Metapenaeus endeavouri; M. ensis*), scampi (*Metanephrops spp.*), Moreton Bay bugs (*Thenus spp.*) and commercial scallops (*Amusium spp.*) (AFMA 2013). The fishery is located off Australia's northern coast, between Cape York in Queensland and Cape Londonderry in Western Australia (Figure 2.1).

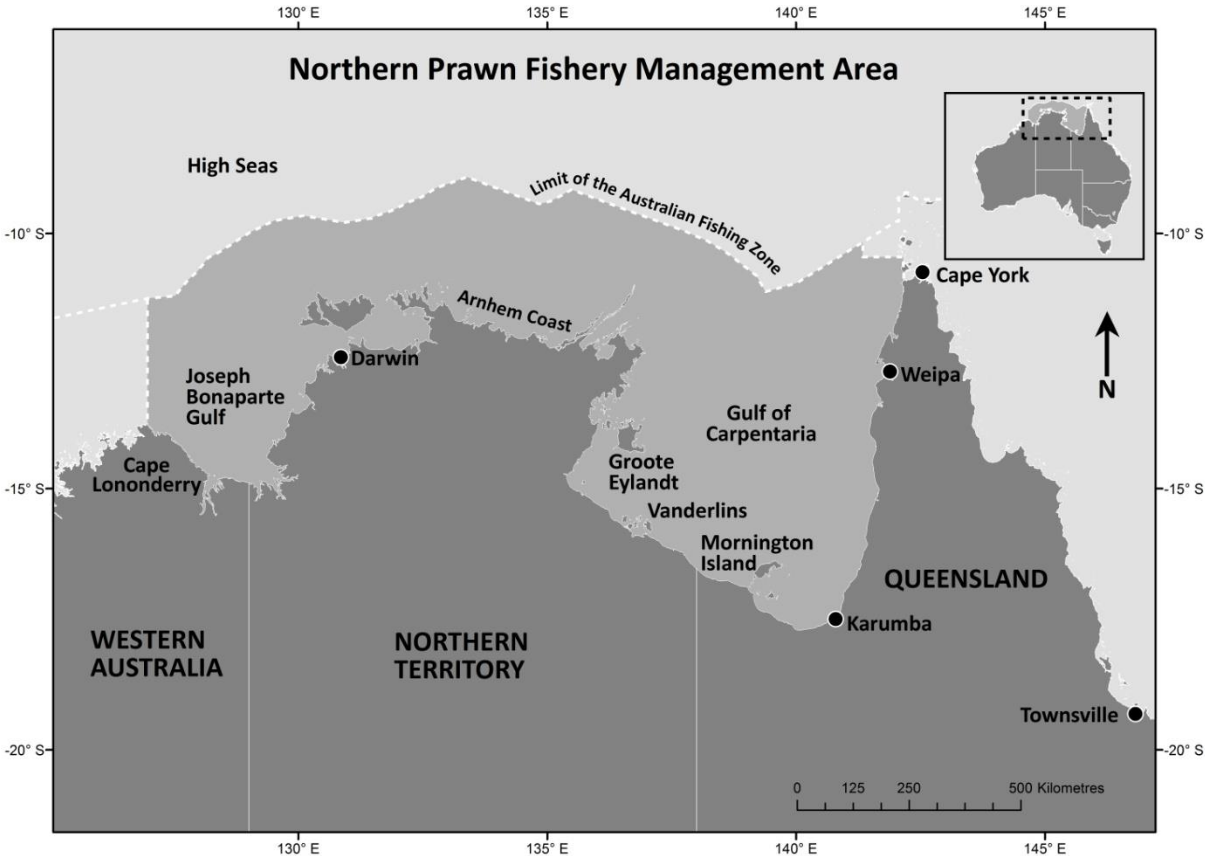

Figure 2.1 The Northern Prawn Fishery, Australia

The white banana prawn fishery is effectively a single-species subfishery within the larger NPF, temporally and spatially separated from the other species (Zhou et al. 2014a). Catch rates of white banana prawns are volatile and heavily affected by environmental conditions, with higher catches generally occurring after wetter than average summers (Vance et al. 2003). The variability of white banana prawn biomass makes it difficult to set appropriate catch or effort limits (Buckworth et al. 2013) and the NPF is managed using input controls implemented under the Northern Prawn Fishery Management Plan 1995 (Barwick 2011). The banana prawn fishery commences when the NPF season opens and usually operates for a few weeks in April/May. Banana prawns are generally caught during daylight hours on the eastern side of the Arnhem Land coast and in Joseph Bonaparte Gulf where the industry use spotter aircraft to identify aggregations to target. More than half of a vessel's daily prawn catch is banana prawns in the banana prawn subfishery. All NPF vessels have catch handling, packing and freezing capabilities and all prawns are frozen at sea. Catch is landed in Karumba or Darwin, or delivered to a mothership, which lands the combined catches from different vessels in Townsville. Prawns are stored frozen before transport to processing.

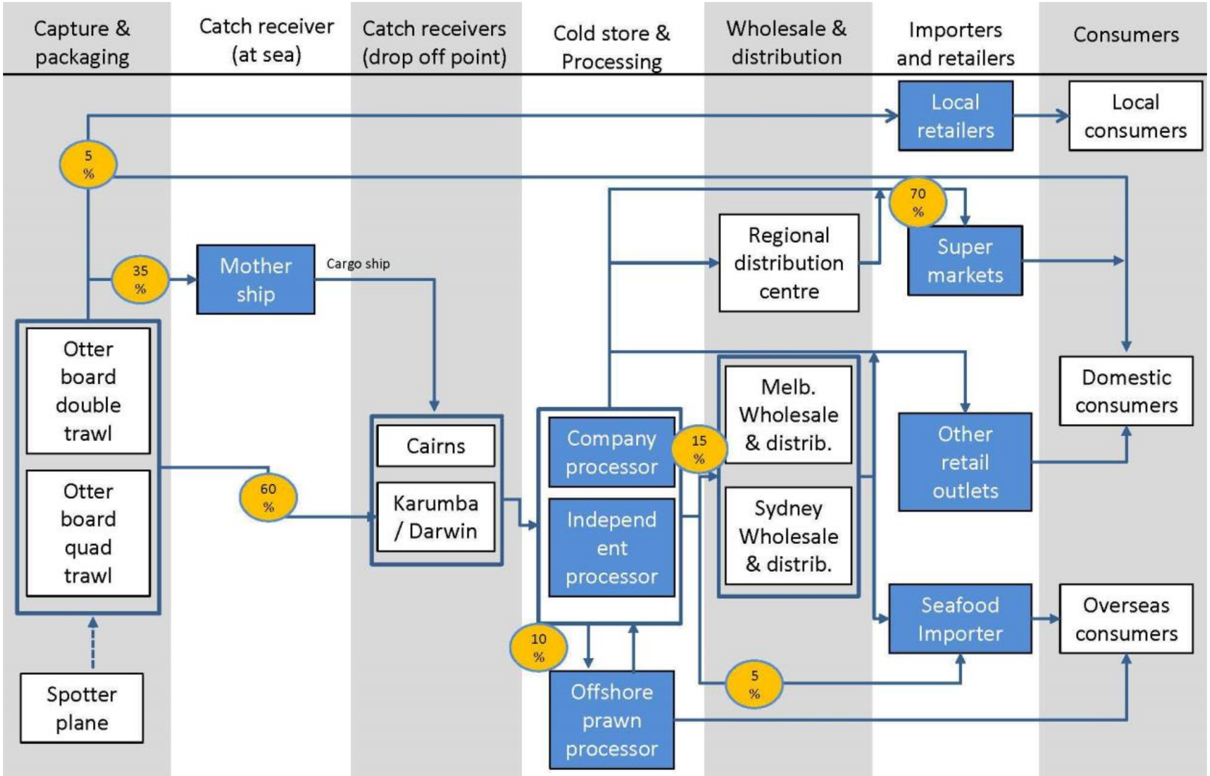

Figure 2.2 Supply chain of banana prawns from the Northern Prawn Fishery

The other main subfishery, the tiger prawn fishery, operates from August to November. Tiger prawns are taken at night and the majority of catch comes from the southern and western Gulf of Carpentaria and along the Arnhem Land coast (Woodhams et al. 2012). More than half of a vessel's daily prawn catch in the tiger prawn subfishery is tiger prawns (Barwick 2011).

The supply chain of the banana prawn is depicted in Figure 2.2 and the following systems were modelled: fishing - including spotter plane, cold-storage, transport to processing, and processing. Under current sustainability assessments, the NPF is assessed as one fishery although distinctions between the two subfisheries are important as they have very different fuel use, bycatch rates and final markets. The majority of the catch from the tiger prawn fishery is exported to Japan via seafreight, while approximately 80 to 90% of white banana prawns are sold on the Australian domestic market (AFMA 2013). An LCA for tiger prawns was not carried out, however the global warming potential (GWP) of the fishing stage and transport to international market stage of the tiger prawn supply chain were modelled for comparison.

2.3.2 Data collection

Data on catch volume, days fished and fuel cost for all vessels operating in the NPF was sourced from the Department of Agriculture, Fisheries and Forestry (Woodhams et al. 2011; ABARES 2011). Fuel cost for each subfishery was not available, therefore the total fuel cost available for the fishery for 2009-11 was allocated across the banana and tiger subfisheries based on the proportion of total days fished in each, consistent with other trawl fishery LCAs (Parker et al. 2014a). Fishers recorded a total of 5031 boat days in the banana subfishery over the two-year period, and 11,228 boat days for the tiger subfishery. Specific catch and financial performance information for the NPF fleet were only available for 2009-11 at the time of this study. As annual catch volume varies in the fishery, sensitivity of results to catch variation was assessed by comparing the life cycle impact results for 1 kilogram of banana prawn using three different scenarios: (a) base case scenario, using catch data and boat days from 2009-2011; (b) 10% increase in catch with the same number of boat days; and (c) 10% decrease in catch with the same number of boat days.

2.3.3 Banana prawn life cycle inventory

The average fuel use per kilogram of banana prawn caught was calculated using data from three sources: (i) for 2011-12 from independent fishers who predominantly use a mothership to land catch, (ii) for 2012 from a company that did not use a mothership, and (iii) for 2009-2010 and 2010-11 from ABARES reports (George et al. 2012; Woodhams et al. 2011) which included a mix of mothership users and non-users. Data from (i) and (ii) were provided in total annual litres and converted to l/kg. Fuel use in litres was calculated for the ABARES data by dividing the total fuel cost for 2009-2010 and 2010-11 by the average price of diesel (Motormouth 2012), minus the rebate of AUD $0.38 per litre.

Fuel use for freezing has not been separated from fuel use for fishing in this study, as freezing occurs on-board the fishing vessels. Fuel use for the spotter plane was provided by a private company and converted to KJ. The abiotic effects of antifoul use, fishing gear and cardboard packaging of frozen prawns were also included for the capture stage as these goods need to be regularly replaced, unlike other capital goods such as the fish boats themselves. The life span of the fishing gear was determined through discussions with fishers and retailers. Refrigerant use on boats and at processing was not included in the LCA due to data availability and the current phase out of the main refrigeration gas, R22, in Australia. The impacts of fuel used for freezing on the boats are captured under the GWP and CED indicators. Truck operation was increased by 22% to account for energy used in freezing equipment (Berlin and Sand 2010). Refrigerant use was captured for cold storage as the data was adapted from Ziegler et al. (2011). Refrigerant use in the NPF is presented in the discussion section.

Processing refers to the activities that occurred within a land-based processing facility. Figures on total water and electricity at processing were sourced from a facility belonging to one of the largest vertically integrated companies operating in the NPF. Processing was minimal and involved thawing in fresh water, grading and repacking. Some of the catch packed at sea is not reprocessed ashore and would therefore require fewer inputs than the prawns examined here. Banana prawns accounted for 30% of the total products processed at the facility and input use was allocated based on mass. Data on cold-storage was calculated based on data reported by Ziegler et al. (2011) for pink shrimp, and relate to land-based cold-storage only. Transport distance from landing to processor via truck, or truck and mothership, was calculated through Google Maps. Capital goods such as fishing boats, vehicles and buildings, were excluded as they are generally of minor importance for LCA (Ellingsen and Pedersen 2004; Thrane 2004b; Ellingsen and Aanondsen 2006; Hospido and Tyedmers 2005). The retail and consumption stages of the supply chain were not included.

2.3.4 Tiger prawns

Fuel use in litres for tiger prawns was calculated by dividing the total fuel cost for 2009-2010 and 2010-11 by the average price of diesel (Motormouth 2012), minus the rebate of AUD $0.38 per litre. Transport distance from Australia to Japan was calculated using Google maps and time taken for refrigerated seafreight calculated using Ports.com.

2.3.5 Life cycle impact assessment

Environmental impacts associated with the capture, storage, processing and transport of banana prawns were evaluated using Life Cycle Assessment (LCA), a holistic method for the standardised assessment of products and production methods along the supply chain (ISO 2006b, a). The functional unit of comparison used was 1 kg of frozen prawn at the processor gate, represented in the results section as kg^{-1} prawn, which is approximately 550 g of prawn meat. Impact categories, or indicators, were selected from the Australian Indicator Set (v2) (Life Cycle Strategies 2012). Life cycle inventory libraries that originate from Europe or the United States are not always relevant for

Australia, therefore locally adapted libraries have been developed to help standardise the interpretation of ISO 14040 in Australia (alcas.asn.au/AusLCI). These Australasian libraries were used where possible, and Ecoinvent libraries used where local data was not available. Impact assessment methods were selected from the Australian impact method available in Simapro 7.

Of the indicators available through the Australian indicator set, global warming potential (GWP), eutrophication potential (EP), water use, cumulative energy demand (CED) and marine aquatic ecotoxity were deemed the most relevant to the systems examined. They also complimented indicators selected for inclusion in the National Life Cycle Inventory (LCI) for agricultural products (Eady et al. 2014). For the GWP indicator, 100-year impacts were based on the Intergovernmental Panel on Climate Change (IPCC) (IPCC 2006). Sensitivity analysis was used to test variation in the GWP and EP results from the locally adapted methods with the methods commonly used in other LCAs: CML (Center for Environmental Studies) 2 baseline 2000 and ReCiPe. The indicator for embodied energy, CED, is included in the Australian indicator set but not in CML or ReCiPe. It was included in this study given that crustacean fisheries are one of the most energy intensive. Total energy flows for CED were based on lower heating values. To ensure marine aquatic ecotoxicity values were relevant to Australian conditions, they were based on Australian characterisation factors and normalisation figures (Lundie et al. 2007) and updated in 2010 based on the consumer price index 2005/6. The measurement unit for this indicator, which reflects the total residence time in water of the active substance (Day), was different to those used in other LCA methods and results were therefore not comparable.

Australia is an arid continent so the water use indicator was included, noting it is less well developed than other indicators (Grant and Peters 2008) and is simply an inventory of the total amount of water used. The normalisation factors for this indicator were taken from the Australian Bureau of Statistics (Australian Bureau of Statistics 2006) and no distinction was made between types of water used (see for example Mekonnen and Hoekstra 2012; Owens 2001). All water use in this assessment refers to unspecified water of natural origin. Ozone layer depletion and photochemical oxidation (smog) indicators are not included in the Australian indicator set, despite their common use in seafood LCAs conducted in the Northern Hemisphere. The use of ozone-depleting substances (ODS) is not significant in Australia, where most ODS emissions are from pre-existing sources such as old equipment or leaking landfills (Fraser et al. 2013), and smog incidents are rare (Grant and Peters 2008). Prawns from the NPF are treated with a 1% sodium metabisulphite solution for the cosmetic discolouration ‘black spot’, however, the contribution of this preservative to ozone layer depletion using the CML method, per kilogram of prawn, is negligible. While ocean acidification is of relevance to seafood systems, it is caused by increased atmospheric CO_2 (Lough and Hobday 2011) and is captured by the global warming indicator.

A number of LCA studies have presented fishery-specific impact categories, such as the global discard index (GDI) (Vázquez-Rowe et al. 2012c), the seafloor impact potential (SIP) (Nilsson and Ziegler 2007), and primary productivity required (PPR), alongside conventional impact categories. Seafloor damage, bycatch and discards were excluded from the life cycle impact assessment but are discussed throughout this study. Primary production can limit global fisheries yield (Chassot et al. 2010) and PPR is an expression of the primary productivity consumed by an organism given its trophic level (TL). This indicator is not currently formalised into LCIA methods and is calculated by an equation developed by Pauly and Christensen (1995).

$$PPR = (catch/9) \times 10^{(TL-1)}$$

The PPR estimate is based on a ratio of 9:1 for the conversion of wet weight to carbon and 10% transfer efficiency per trophic level (TL). TL for prawns was taken from the Seas around us project (www.seasaroundus.org).

2.4 Results

2.4.1 Banana prawns

The global warming potential (GWP) of one kilogram of frozen whole white banana prawn for the supply chain examined was 7.2 kg CO_2e (Table 2.1, Figure 2.3). The fishing stage was the source of 63%, or 4.3 kg CO_2e kg^{-1} prawn of this GWP. About 60% of the emissions at capture were due to the operation of the trawl vessel engine which uses fuel for fishing and freezing. The use of a spotter plane made a negligible contribution to the GWP, as did on-board packaging and the antifoul used on the boats. Despite weighing in excess of 1000 kg per boat, the trawl gear of steel otter boards contributed only 0.03 kg CO_2e kg^{-1} prawn, due to their repeated use over time. The transport stage made little contribution to any indicator measured. GWP of transport was less than 4% or 0.3 kg CO_2e kg^{-1} prawn (Figure 2.3), for either a journey of 2,800 km by refrigerated truck from Far North Queensland to the south of the state, or a 1,700 km journey on a mothership, followed by 1,500 km from Cairns to Brisbane by refrigerated truck.

Fuel use by fishing vessels was the main source of cumulative energy use, and was closely aligned to the GWP, accounting for 60% of the CED indicator or 63 MJ kg^{-1} prawn. Storage and processing together accounted for 37.41 MJ kg^{-1} prawn. Fuel use was also the main contributor to the eutrophication potential indicator, accounting for 86% of EP or 0.01 kg PO_4e kg^{-1} prawn (Table 2.1, Figure 2.3). Diesel fuel used on the fishing vessels contributed to eutrophication through the production of nitrogen oxides.

Table 2.1 Life cycle impact assessment results for 1kg of frozen banana prawn (2009-2011)

LCA stage	Process	Global warming potential (kg CO_2e)	Eutrophication potential (kg PO_4e)	Cumulative energy demand (MJ)	Water (L)	Marine aquatic ecotoxicity (Day)
a. Capture	Boat engine	4.20	9.93E-03	61.42	0.47	1.84E-11
	Aircraft engine	2.61E-05	1.42E-08	3.93E-04	2.98E-06	1.19E-16
	Packaging (cardboard)	0.05	5.03E-05	0.92	1.61	3.18E-12
	Antifoul	1.49E-03	2.16E-05	0.03	0.08	4.51E-11
	Gear	0.03	1.74E-05	0.65	0.05	9.31E-13
Sub total		**4.29**	**0.01**	**63.03**	**2.20**	**6.76E-11**
b. Storage	Freezer	1.43	8.00E-04	20.30	3.60	2.49E-10
c. Processing	Water	0.01	4.33E-06	0.14	15.91	9.68E-13
	Electricity	1.19	6.71E-04	16.98	2.97	2.08E-10
Sub total		**2.64**	**1.48E-03**	**37.41**	**22.47**	**4.58E-10**
d. Transport	truck/ mothership	0.27	1.27E-04	3.96	0.03	1.20E-12
Total		**7.20**	**0.01**	**104.40**	**24.71**	**5.27E-10**

The processing stage accounted for the largest share of total water use, 76% or 16 L kg^{-1} prawn. The fishing stage contributed less than 10% to water use, although the main source of water consumption for this stage was cardboard packaging. Processing and cold-storage together accounted for 87% of marine aquatic ecotoxicity, due to emissions from the use of coal-fired electricity in Queensland. Antifoul accounted for less than 10% of total ecotoxicity, however, at capture it was the source of 67% of the ecotoxicity.

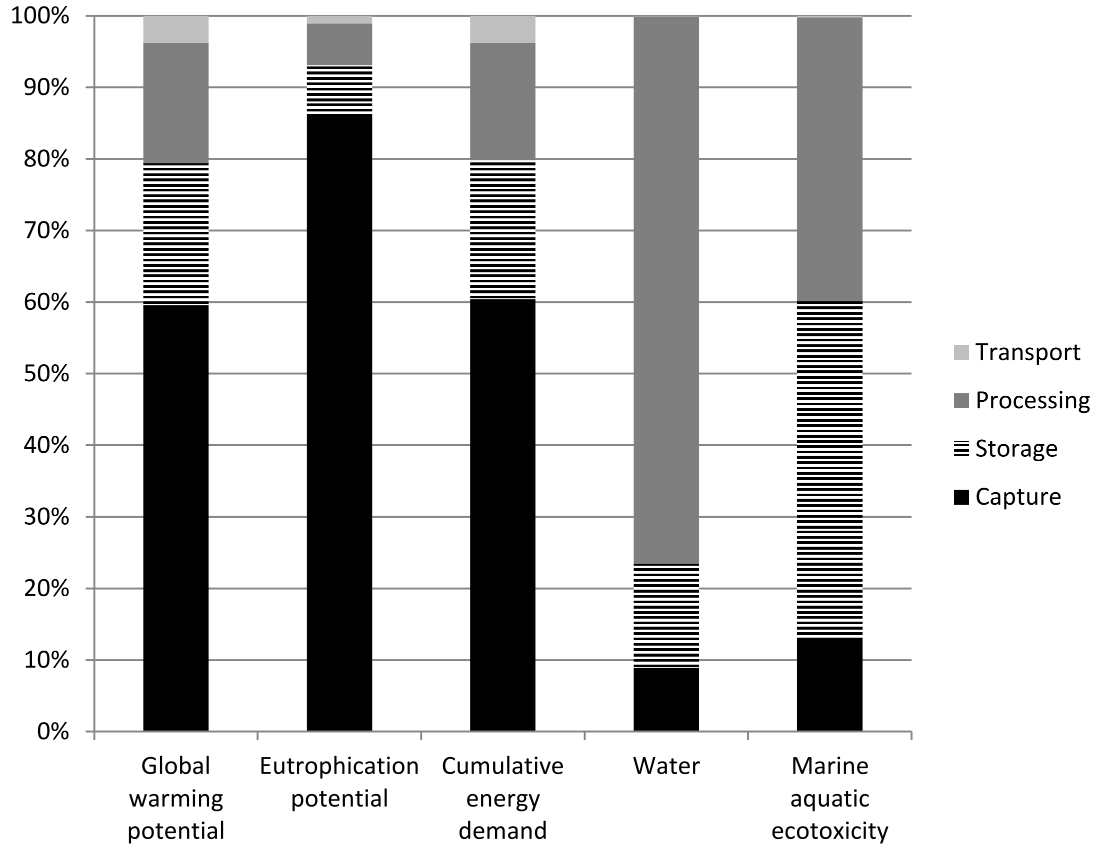

Figure 2.3. Life cycle impact assessment of 1 kg banana prawn (2009-2011)

The banana prawn has a trophic level (TL) of 3 and following the equation developed by Pauly and Christensen (1995) the PPR was calculated at 11.1 t C kg-1 for landed banana prawns.

2.4.2 Tiger prawns

Tiger prawns required comparatively more fuel than banana prawns and therefore had much higher fuel use intensity (FUI) per kilogram. The GWP of the fuel used to catch tiger prawns was 7.6 times greater than banana prawns per kilogram, equating to almost 28 kg CO2e kg^{-1} prawn (Table 2.2). The majority of tiger prawns are exported to Japan and the GWP for this transport stage was 13.6 kg CO2e kg^{-1} prawn by airfreight or 0.33 kg CO2e kg^{-1} prawn by sea freight, noting that most are shipped by sea. In contrast, the footprint of 1 kg of banana prawn sold in Australia was 0.019 kg CO2e kg^{-1} prawn for transport of 100 km or 0.77 kg CO2e kg^{-1} prawn for transport of 4,000 km by refrigerated truck to retail.

Table 2.2 Fuel use intensity and carbon emissions for 1 kg banana and tiger prawn (2009-2011)

	Fuel use intensity (l/kg)	Carbon footprint (kg CO_2e) from fuel use at fishing	Main retail destination	Distance from processor to retail (km)	Carbon footprint (kg CO_2e) from fuel use in transport
Banana prawn	1.3 ± 0.3	4.2	Australia	a. 100 b. 4000	a. 0.02 b. 0.77
Tiger prawn	9.9 ± 0.5	32	Japan	8200 (sea freight) 7800 (air freight)	0.33 13.6

2.4.3 Sensitivity and scenario analyses

Sensitivity of results to choice of impact assessment method was small. For climate impacts under the GWP indicator, results differed by less than 0.1 kg CO2e kg^{-1} prawn between different methods. Results for EP were identical using the Australian indicator set or CML, as EP was quantified according to the Centre for Environmental Studies (CML) 2 Baseline 2000 method (CML 2001). The ReCiPe method used a different unit of measurement (kg N equivalent) and was therefore not comparable.

Table 2.3 Modelled changes in emissions and energy use per kilogram frozen prawn as a result of potential changes in annual catch in the NPF

Scenario	**Resulting change in impacts (%)**				
	GWP	EP	CED	Water	Ecotox
10% catch increase	-5	-7	-5	0	0
10% catch decrease	7	9	7	0	0

Increasing catch through scenario analysis resulted in potential improvements across most indicators, including EP, GWP and CED, as a result of improved FUI. Decreased catch, in contrast, resulted in potentially higher emissions and energy use (Table 2.3) per kilogram of prawn. Under these scenarios the capture stage remained the main source of impacts for the supply chain modelled. Ecotoxicity and water use were not sensitive to changes in catch rates as they were predominantly influenced by processing.

2.5 Discussion

The supply of prawns to domestic and international markets is responsible for a range of impacts, including GHGe, resource use, eutrophication and ecotoxicity that are not currently considered at a fisheries or supply chain level, yet have the potential to impact the fishery and the environment more broadly.

2.5.1 Broadening the scope of seafood sustainability assessments

In terms of resource use, trawl fishing gear is routinely more energy intensive than other gear types (Tyedmers 2004) and fuel use in prawn trawling in particular is typically greater than in other fisheries (Tyedmers et al. 2005; Smith 2007). The cumulative energy demand of banana prawns and associated carbon footprint, based on kilograms of CO_2 equivalent emissions, is lower than that of tiger prawns from the same fishery, in part due to the aggregations that banana prawns form, which make them easier and more efficient to target. Fishing for tiger prawns is more fuel intensive as they are more dispersed and do not congregate in boils, requiring boats to trawl longer hours for lower catch. The GWP of tiger prawns in the NPF is similar to that of trawl caught pink shrimp, 29 kg CO_2e kg^{-1} shrimp (Ziegler et al. 2011). Aggregating behaviour is rare in penaeids but, in species in which it does occur, is strongest at high stock levels (Die and Ellis 1999). The GWP of the banana prawn fishery is therefore strongly linked to fishery management and allowing stock abundance to fall may reduce catchability and increase GHGe through reduced fuel efficiency.

Fishery management, in the form of capacity reduction programmes over the past decade, has led to decreased fuel use by boats in the NPF (Pascoe et al. 2012), and presumably reduced GHG emissions although these have not been monitored over time. Fuel use intensity (FUI) has fallen from 3 L kg^{-1} for banana prawns in 2006 to 1.5 L kg^{-1} in 2010 and from almost 11 L kg^{-1} for tiger prawns in 2006 to around 7 L kg^{-1} in 2010 (ABARES 2011). This reduction is attributable to changes in fishery management as well as technological and behavioural changes in fishing businesses, which were driven by external forces such as the increasing cost of fuel. Monitoring GWP relative to catch into the future may be valuable. For example, recent attempts to adjust exploitation levels in this fishery to maximum economic yield, in response to negative changes in economic conditions and the expected trajectory for the fishery (Pascoe et al. 2013), has the potential to further improve FUI and reduce GWP (Chapter 3, Farmery et al. 2014).

Seafood LCA studies that include the processing stage of the life cycle are limited (Vázquez-Rowe et al. 2012a). Processing can constitute a key contributor to the potential environmental impacts for seafood products, particularly for more complex processing including canning (Iribarren et al. 2010a; Thrane et al. 2009; Almeida et al. 2015; Avadí et al. 2015; Avadí et al. 2014) and packaging in stand-up pouches (Mungkung et al. 2006). The results presented in this chapter were consistent with a Danish study where processing of frozen prawns represented a relatively small overall impact, yet consumed large amounts of water in comparison to other stages (Thrane 2004b). Water use for banana prawn at the processor gate was 25 L kg^{-1} whole prawn, equating to 45 L kg^{-1} prawn meat (assuming a 55% recovery rate). This figure is comparable with other Australian seafood, 71 L kg^{-1} of lobster meat (Chapter 3, Farmery et al. 2014) and 61 L kg^{-1} of fish meat from the Commonwealth Trawl Sector (Appendix 1). Water use indicators are rarely included in LCAs of food products (Koehler 2008) despite the high water-intensity of animal products (Pimentel et al. 2004). Water use for seafood production is low in the context of food production, for example in comparison to global average beef production at 15 415 L kg^{-1} (Mekonnen and Hoekstra 2012), which may be an important consideration given that water availability is likely to limit future food production (Hanjra and Qureshi 2010), particularly in arid countries like Australia.

Processing of banana prawns was also the source of GHGe and ecotoxicity impacts. In this study, the provision of coal-powered electricity for cold-storage and processing was a greater source of aquatic toxicity than fishing. Previously, ecotoxicity in seafood LCAs has been associated with the use of antifoul (Ziegler et al. 2011) and burning of diesel fuel on fishing boats (Vázquez-Rowe et al. 2010b). Most processing involving product transformation of prawns from the NPF occurs outside Australia and the environmental footprint of banana prawns consumed in Australia is therefore dependent on the type of processing undertaken and the mode of transport used if processed offshore. Airfreight was shown in this study to dramatically increase the GWP of tiger prawns exported to Japan, which is consistent with other LCA studies (Chapter 3, Farmery et al. 2014; Andersen 2002; Winther et al. 2009).

Primary Productivity Required (PPR) is increasingly used in assessments of seafood sustainability, where it serves as a measure of biological resource use from aquaculture or fisheries (Hornborg et al. 2013). This is of importance for some specific fisheries where the rate of biomass removal, in terms of PPR, exceeds the limits required for long-term sustainable marine ecosystem production (Coll et al. 2008). Banana prawns have a relatively low trophic level and therefore appropriate less primary productivity per kilogram than other commercially caught seafood eaten in Australia, including

Gould's squid (*Nototodarus gouldi*) TL=3.5 PPR= 35.1 t C kg^{-1}, yellowfin tuna (*Thunnus albacares*) TL= 4.34 PPR=243.1 t C kg^{-1}, and tiger flathead (*Neoplatycephalus richardsoni*) TL= 3.9 PPR = 88.3 t C kg^{-1}. Lower PPR values are associated with lower ecosystem costs, however, further research is needed to progress this indicator and standardise its use for quantifying ecosystem effects of fishing (Avadí and Fréon 2013).

Refrigerant leakage increases the GWP of seafood between 13 – 20% (Vázquez-Rowe et al. 2012b, 2010b; Iribarren et al. 2011) and reducing such leakage therefore presents the potential to reduce GWP in many fisheries. Data on refrigerant leakage across the NPF supply chain was not available, however, prawns are rapidly frozen at sea on fishing vessels and stored at -35˚ C, sometimes for weeks before unloading. R22 is the most commonly used refrigerant in the NPF, on fishing trawlers, in processing factories and in cold-storage facilities (NPF Industry 2014). This refrigerant has a climate impact indicator of 1810 kg CO_2e kg^{-1} (IPCC 2007) and is currently being phased out in Australia. Fishers are therefore looking to new concepts in refrigeration in existing boats and new trawler designs. A recent report on refrigeration technology options for the Northern Prawn Fishery fleet reported that HFC 507A was the only gas suitable to replace R22 (Expert Group 2013). While this replacement gas does not deplete ozone, it has a much higher GWP of 3985 kg CO_2e kg^{-1} (The Climate Registry 2014). The GWP of prawns from the NPF will likely increase following the transition of the NPF trawlers and associated cold-chain from R22 to HFC 507A, assuming all other factors remain the same. The contribution of HFC to climate change has been recognised as an unintended negative side effect of actions to limit ozone depletion (Velders et al. 2012).

2.5.2 Integrating new LCA indicators with current sustainability assessments

The NPF was one of the first Australian fisheries to assess the ecological sustainability of its supporting ecosystem through ecological risk assessment (Zhou and Griffiths 2008), the same framework that underpins MSC certification (Hobday et al. 2011). The fishery is managed to meet the goal of Ecologically Sustainable Development (ESD) and has been accredited under the EPBC Act 1999 as environmentally sustainable (FRDC 2012). It has also been recognised by the Food and Agriculture Organization of the United Nations as a global model for fisheries management (Gillett 2008) and has recently received independent third-party accreditation under the Marine Stewardship Council's Certification program for banana and tiger prawns (Pascoe et al. 2013). The fishery was assessed against the MSC standard which is based on three over-arching principles; viability of the target stock, impact on the ecosystem and management of the fishery. The NPF is one of only eight prawn fisheries worldwide that have attained the MSC global standard by meeting the internationally-recognised environmental standards (MSC 2012).

Prawn trawling, particularly in tropical regions, is responsible for some of the highest rates of bycatch and discards recorded in wild capture fisheries (Dumont and D'Incao 2011; Stobutzki et al. 2001; Eayrs 2007). Several LCA studies have included biological indicators to quantify these impacts (for details of published fisheries LCAs using these indicators see Vázquez-Rowe et al. 2012a; Avadí and Fréon 2013), however, the indicators are yet to be standardised. Results are typically presented as kilograms per functional unit (Ziegler et al. 2011; Vázquez-Rowe et al. 2010b) and efforts continue to progress this type of indicator in order to better understand the specific environmental impacts (Vázquez-Rowe et al. 2012c). Bycatch in Australia has been estimated at 25% of total catch of trawl fisheries (Davies et al. 2009). Bycatch varies greatly between banana and tiger prawns in the NPF, as banana prawns have a higher mean bycatch catch rate but lower total bycatch than the longer duration trawls of the tiger prawn fishery (Dell et al. 2009; Zhou and Griffiths 2008). Bycatch in the NPF comprises between 87.5% and 95.2% of the total catch of the fishery (Pender 1992; Brewer et al. 2006), most of which is returned to the sea either dead or dying (Brewer et al. 2007; Pender 1992). Spatial variation in the fishery has been recorded with a bycatch-to-prawn ratio of 0.8:1 in the Joseph Bonaparte Gulf region of the NPF (Dell et al. 2009) and 5:1 in the Gulf of Carpentaria region (Tonks et al. 2008). Bycatch in the Southern Pink Shrimp fishery in Senegal was similarly high, with fish representing 88% of landings by mass and 77% of bycatch discarded (Ziegler et al. 2011). In contrast, discards represented only 3.9% in the Peruvian anchoveta small- and medium-scale fishery (Avadí et al. 2014).

Management actions in the NPF, including compulsory use of a specific suite of Turtle Excluder Devices (TEDs) and Bycatch Reduction Devices (BRDs), have reduced the fishery's impact on bycatch (Brewer et al. 2006; Heales et al. 2008; Burke et al. 2012). An Ecological Risk Assessment with management arrangements for bycatch species and a bycatch and discard action plan has also been implemented in the fishery (Barwick 2011). The inclusion of a bycatch or discard indicator, such as the GDI proposed by Vázquez-Rowe et al. (2012c), in future fishery LCAs could help evaluate the effectiveness of such plans and management changes. Possibly the most significant management change in the fishery that has affected bycatch has been the reduction in effort and fleet size, from over 300 vessels to the current fleet of 52 (Barwick 2011).

The use of bottom-trawl gear in fisheries disturbs seabeds, potentially leading to substantial changes in benthic community structure and habitat (Kaiser et al. 2006; Althaus et al. 2009; Williams et al. 2010; Gislason et al. 2000; Collie et al. 2000). Much of the research in this field has taken place in the northern hemisphere where the impacts have often been substantial (Thrush and Dayton 2002; Heath and Speirs 2012). The results of such studies have strongly influenced the perceptions of trawl fishery impacts (Dichmont et al. 2013) and resulted in actions such as the proposed phase-out of deep-sea bottom trawling and bottom gillnet fishing by the European Commission (PEW 2013). Trawling in tropical and subtropical regions of Australia has local and specific impacts, particularly where fishing grounds overlap with vulnerable biota (Williams et al. 2010; Pitcher et al. 2009a; Svane et al. 2009). Substantial variation exists in seafloor impacts by trawling, however, and Burridge et al. (2006) found that trawling in Northern Australia did not have a major impact on the demersal fauna. Other authors have shown that trawling is benign on habitats where the benthos is resistant to trawling (van Denderen et al. 2013), or even beneficial to the fishery, where it may increase production of some fish species (van Denderen et al. 2013; Rijnsdorp and van Leeuwen 1996).

An indicator of seafloor impact potential (SIP) proposed by Nilsson and Ziegler (2007) has been trialled in LCA to measure the amount of seafloor dragged by trawlers and other gear, noting that the SIP for other fishing methods typically amounts to zero (Vázquez-Rowe et al. 2012c; Ziegler et al. 2003). The swept seabed area is calculated by multiplying trawl effort by area swept per hour and results are typically presented as km^2 per functional unit (Ziegler et al. 2011; Vázquez-Rowe et al. 2012c, b). Results are more meaningful however when overlayed with habitat maps to determine fishing pressure in sensitive habitats and recoverability potential, and when concentration of fishing effort is calculated to determine actual area affected by trawling, as described by Nilsson and Ziegler (2007). In the NPF, fishing takes place in depths shallower than 40 metres and it is estimated that less than 10% of the total area is trawled (Zhou and Griffiths 2008). 2.1% of the total area is never trawled due to permanent area closures, including all shallow water seagrass. Areas that are unsuitable for trawling, such as large reef outcrops and areas with low density of the target prawn species, are also not trawled (AFMA 2013). Of the area that is trawled, some is reportedly unconsolidated sediments that are resilient to perturbation by trawl gear. While the impacts of sparse and infrequent trawl effort are not currently considered a threat to biodiversity in the NPF (Pitcher et al. 2009a), the correlation between fishing effort and potential effect on seafloor (Vázquez-Rowe et al. 2012c) suggest that alteration of current gear configurations and fishing intensity could result in greater impacts. The SIP indicator could therefore be used in future LCAs of the NPF, in combination with the method presented in Chapter 5, to track these types of changes.

2.6 Conclusions

Expanding the scope of environmental considerations in the NPF, by incorporating standardised and emerging life cycle indicators, could enhance current assessments of seafood sustainability and offer new insights into fisheries management. For example, in this chapter I identified that LCA could be used to monitor improvements in the fishery and other supply chain stages though time, and to highlight differences between fisheries, or between species within the same fishery. Opportunities for LCA to complement existing fisheries management include the inclusion of a bycatch or discard indicator, inclusion of a seafloor indicator and monitoring of GWP. Reductions in impacts assessed through LCA may complement the achievement of other management targets, as illustrated by the indirect reduction in GHGe that have been occurring as a result of improved efficiency in the fishery. In cases where management actions and GHGe do not move together, where there are fuel subsidies for example, life cycle indicators are needed to capture the trade-offs. NPF stakeholders across the supply chain stand to benefit from the demonstration of targeting broader sustainability goals, through an advantage in a market where consumers are increasingly aware of, and willing to pay for sustainability (Macfadyen and Huntington 2007). Furthermore, the inclusion of important LCA indicators, such as GWP, as an integral part of existing fishery assessments, is a strategic move in adapting to an increasingly carbon-regulated world.

CHAPTER 3: LINKING LIFE CYCLE ASSESSMENT AND THE MANAGEMENT OF MARINE RESOURCES

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3.1 Abstract

The concept of seafood sustainability does not typically include the energetic or material demands of the capture or supply chain processes, despite the significant impacts they generate. Life cycle assessment (LCA) was used to measure the environmental footprint of the supply of Tasmanian southern rock lobster, *Jasus edwardsii* (TSRL). International airfreight of live lobsters was the major contributor to global warming potential (GWP) and cumulative energy demand (CED) indicators, while the fishing stage accounted for the majority of impacts to eutrophication potential (EP), water use and marine aquatic ecotoxicity. The environmental footprint of the TSRL in the scenarios modelled was responsive to marine resource management decisions made inside and outside the fishery. Targeting maximum economic yield rather than maximum sustainable yield decreased the carbon footprint by 80% or 10 kg CO_2e kg^{-1} of lobster at capture. Limiting access to the fishery by increasing the coverage of marine protected areas increased the fishery's carbon footprint by 23% or 3 kg CO_2e kg^{-1} of lobster at capture. The unintended consequences of management changes suggest that in a future of increased carbon emission regulation, marine resource decision-making should not be made in isolation of broader environmental impacts.

3.2 Introduction

This chapter builds on the findings of Chapter 2 by further examining the relationship between fisheries management and LCA. Improving the environmental sustainability of seafood supply is typically associated with protecting the target species (Worm et al. 2009), non-target species (Hilborn 2007b) and reducing ecosystem impacts (Pelletier and Tyedmers 2008), as fisheries management evolves towards an ecosystem-based fisheries management (EBFM) approach (Zhou et al. 2010). However, the broader environmental impacts generated by fisheries, in particular the use of fossil fuel in vessels (Tyedmers et al. 2005; Tyedmers and Parker 2012; Ziegler and Hansson 2003; Thrane 2004a) and the transportation of landings (Andersen 2002; Winther et al. 2009; Karlsen and

Angelfoss 2000), have largely been excluded from the ecosystem approach, despite their substantial impact (Pelletier et al. 2007).

The implications of improving our understanding and management of the wider impacts of seafood production are significant given the scale of global seafood production. In 2011 approximately 154 million tonnes of seafood was produced globally from capture fisheries (marine and inland) and aquaculture (FAO 2012), accounting for approximately 7% of all protein consumed (FAO 2011). Food production is expected to increase due to growing demand (FAO 2009a), with demand for animal protein in particular influenced by the growth in affluence of emerging economies (Speedy 2002).

Marine capture fisheries contributed 51% of the total seafood produced in 2011 (FAO 2012) and at the same time were accountable for about 1.2% of global oil consumption and the emission of more than 130 million t of CO_2 into the atmosphere (Tyedmers et al. 2005). Additional emissions are generated by processes occurring beyond the capture phase in marine fisheries, in particular from transport, as seafood is the most highly traded food product (Smith et al. 2010). Over 5% of the world annual seafood catch is transported by air freight and this figure will likely increase with growing demand for fresh fish (FAO 2013).

Fisheries are managed for a range of objectives, encompassing biological, economic, social and political goals (Hilborn 2007a). Harvests can be controlled by many methods, broadly grouped as either input or output (catch) controls (Beddington et al. 2007). Output controls directly limit the amount of fish which can be taken from the water each period with a Total Allowable Catch (TAC). Input controls indirectly control the catch through restrictions on fishing, such as limits on the number of licences, capacity of boats, and gear restrictions.

A common historic goal for sustainable harvest in fisheries is Maximum Sustainable Yield (MSY) (Worm et al. 2009), where ongoing biological yield, or food production, is maximised (Figure 3.1). This objective can be implemented by applying the level of fishing effort that produces the maximum yield, without affecting long-term productivity (Sparre and Venema 1998). MSY has been incorporated into the 1982 United Nations Convention on the Law of the Sea, thereby facilitating its integration into national fisheries acts and laws in several countries (Mace 2001). While MSY provides maximum sustainable biological production, it does not necessarily maximise other common objectives such as employment, ecosystem preservation or economic profitability (Hilborn 2007a; Larkin 1977; Punt and Smith 2001; Mardle et al. 2002).

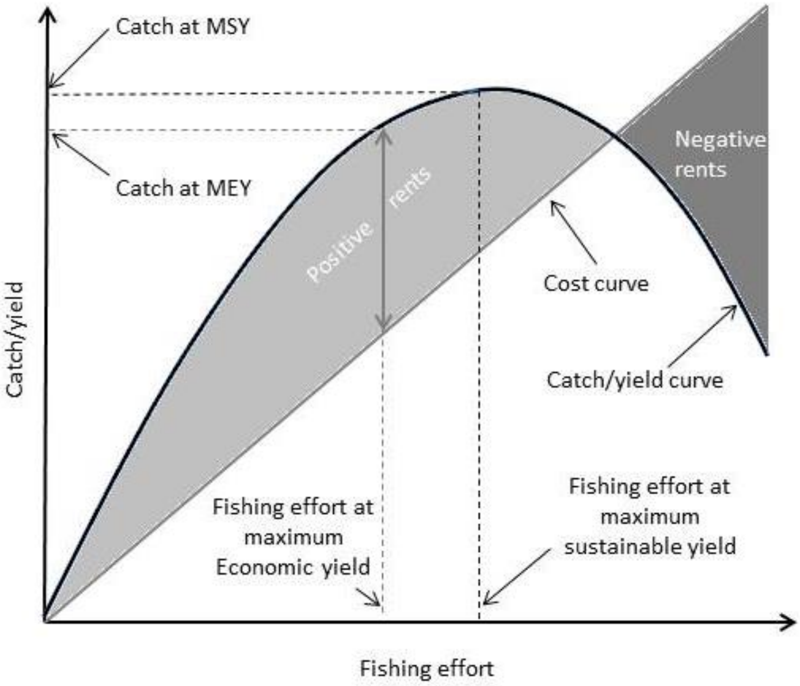

Figure 3.1. Relationship between Maximum Sustainable Yield (MSY) and Maximum Economic Yield (MEY), based on the original Schaefer model as presented by the World Bank and FAO (2008).

Sustainable Maximum Economic Yield (MEY) has recently been implemented as an alternative fisheries management target (Grafton et al. 2010) including in many fisheries in Australia and the United States. Under a MEY harvest target, economic yield is maintained sustainably over the long run at the biomass or effort level where the difference between the costs of harvesting the fish and the revenues obtained from the catch is greatest (Figure 3.1) (Norman-López and Pascoe 2011). While there are varying levels of acceptance of MEY (Christensen 2010), it is generally agreed that compared to a MSY-managed fishery, a target of MEY tends to be more conservative and will generally result in reduced fishing mortality (or catch) and higher biomass (Kompas et al. 2011). This occurs because economic yield is affected by the cost of fishing, which is reduced when biomass or stock abundance is higher.

Objectives related to sustainability of the marine environment are also targeted directly through management systems, for example, Marine Protected Areas (MPAs), which aim to protect biodiversity (Browman and Stergiou 2004). MPAs can affect commercial fisheries and assessments of the impacts of closing areas to fishing typically account for effects on catch and profit in the fishery, but not the effects on the broad environmental impacts of fishing.

Life cycle assessment (LCA) was used to examine the unintended, and generally unacknowledged, environmental consequences of commonly applied fishery management policies and a competing marine resource use on the footprint of supplying Tasmanian southern rock lobster (TSRL) for export. LCA is a tool endorsed by the United Nations to promote sustainable patterns of production and consumption, and to increase the eco-efficiency of products and services (Hertwich 2005). This research illustrates how incorporating LCA considerations into fisheries management can provide information required to enhance the sustainability of seafood supply.

3.2.1 Study fishery: Tasmanian southern rock lobster (TSRL)

Southern rock lobster (*Jasus edwardsii*) was selected as a case study as it is representative of the growing trade in airfreighted seafood and is a single species fishery that experiences a range of management strategies across the 13 jurisdictions where it occurs. The Tasmanian fishery is managed as one stock and commercial catch is taken from areas all around the state. The catch is mainly exported live and marketed to China's growing middle class (ABARE 2009). The TSRL fishery is an inshore coastal fishery, ranging from zero to 200 m depth, where 80% of traps are set at less than 50 m. In the 2010/11 season 236 licensed vessels reported catches of rock lobster (Hartmann et al. 2012). Commercial harvests of TSRL are controlled with a quota management system plus size limits, season and gear restrictions (Gardner et al. 2011). Fishers use baited traps with approximately equal parts Tasmanian caught Australian salmon (*Arripis trutta*) and jack mackerel (*Trachurus declivis*), and barracouta (*Thyrsites atun)* imported from New Zealand.

3.3 Methods

3.3.1 Life Cycle Assessment

Life Cycle Assessment (LCA) provides a holistic framework for comparing products, production methods or changes made along the supply chain using methods standardised through the International Organization for Standardization (ISO 2006b, a). The functional unit of comparison used here was 1 kg of live lobster at the point of arrival in the main export market of Beijing, China. The life cycle includes capture, storage, packaging and transport of live lobsters to market (further information on the supply chain is available in van Putten et al. 2015). The supply chain was included to determine the relative importance of the fishery stage to the environmental footprint under different fishery management scenarios. All processes during capture and export of TSRL were included, however, capital goods such as fishing boats, vehicles and buildings, were excluded as they are generally of minor importance (Ellingsen and Pedersen 2004; Thrane 2004b; Ellingsen and

Aanondsen 2006; Hospido and Tyedmers 2005). Bycatch and discards in rock lobster fisheries are low (Gardner et al. 2011; Brock et al. 2007), including for TSRL juveniles which can exit through mandatory escape gaps in traps, and so the fishery was considered as a single species fishery. While processors occasionally handle other species from other fisheries, the volume of these species is small and does not alter the functioning of the processing facility.

3.3.2 Software and impact assessment methods specific to Australia

Impact categories, or indicators, were selected from the Australian Indicator set (v2) for their relevance to Australia as well as their comparability with other food production LCAs (Life Cycle Strategies 2012). Impact assessment methods from the Australian databases were used to reflect the environmental conditions relevant to the technical systems where they operate. Greenhouse impacts under the global warming potential (GWP) indicator are 100 year impacts based on the IPCC (IPCC 2006) and cumulative energy demand (CED) is the total energy flows based on lower heating values. The eutrophication potential (EP) was quantified according to the CML 2 Baseline 2000 method (CML 2001) and fresh water use and marine aquatic ecotoxicity values were taken from Lundie and Huijbreghts (Lundie et al. 2007) then increased in line with the consumer price index up to 2005/6. Australia is an arid continent so the water use indicator was included although noting it is less well developed that other indicators (Grant and Peters 2008). The normalisation factors for this indicator were taken from the Australian Bureau of Statistics (Australian Bureau of Statistics 2006) and no distinction is made between types of water used (see for example Mekonnen and Hoekstra 2012). All water use in this assessment refers to unspecified water of natural origin. Ozone layer depletion and photochemical oxidation (smog) indicators were not included as the use of ozone-depleting chemicals is not significant in Australia and smog incidents are rare (Grant and Peters 2008). While ocean acidification is of interest to seafood systems, it is caused by increased atmospheric CO_2 (Lough and Hobday 2011) and is captured by the global warming indicator. We have not included ecological indicators in this study, although we note several are available to use from LCA methods and IndiSeas (http://www.indiseas.org/).

3.3.3 Fisheries Management Scenarios

The environmental consequences associated with the biophysical and energetic inputs of fisheries are examined using six management scenarios. These scenarios represent different harvest targets, use of input controls and resource access commonly applied in fisheries management to meet different goals of sustainability (Table 3.1). Each scenario is represented by a rate of landings or catch per unit effort (CPUE) that is either targeted by management in the TSRL fishery (scenarios 1, 2 and 5), and/or is the observed result of management in a southern rock lobster fishery elsewhere adopting the relevant target or policy (scenarios 2,3, 4 and 6).

Scenario 1 reflects the situation in the TSRL fishery in 2010/2011 and therefore incorporates, among other input controls, a limit of 50 traps per vessel which was established prior to TAC management to control total effort. Although the TSRL fishery is quota managed, the emphasis in management up until 2011 had been on achieving higher catch rather than higher economic yield (Gardner 2012). As a result, the catch rate of 0.79 kg / trap lift in 2010/11 was interpreted as an observed catch rate under management targeting MSY. Scenario 1 serves in this analysis as the base case.

Scenario 2 retains the MSY harvest target but allows for the removal of the existing constraint on the number of traps allowed per vessel (Table 3.1). It was assumed the removal of the trap limit would lead to a doubling of traps per vessel, as occurred with vessels operating in the adjacent jurisdiction of Victoria, Australia, when this control was relaxed (Walker et al. 2012) (Table 3.1). In the Victorian fishery, the well capacity of vessels was not a constraint so it was assumed that changing the trap limit would not result in additional trips and fuel use. This assumption was also made on the basis that the fishery is managed with a TAC and thus catch rate (CPUE) would not be affected; that is, the same total number of trap sets would be required to take the same total catch but shared between fewer vessels. Increasing the number of trap sets per vessel has negligible impact on total fuel use per trip because travel between traps is typically around 100 m, which is small compared to travel between port and fishing grounds, often >200 km.

Scenario 3 represents a shift in the fishery to an MEY harvest target while retaining the existing trap limit per vessel. The catch rate value for this scenario was the observed catch rate in the nearby southern rock lobster fishery of CRA8 in New Zealand, which has pursued MEY management objectives since the late 1990s (Miller and Breen 2010). The result has been an increase in CPUE from less than the current Tasmanian rate to 3.8 kg / trap lift (NZRLIC 2011). This high catch rate appears biologically feasible in Tasmania given it is less than historic catch rates, which exceeded 4 kg / trap lift with less sophisticated equipment, in the 1950s (Hartmann et al. 2012). In scenario 4 the catch rate also reflects an MEY target in the fishery plus the trap limit is abolished.

Table 3.1 Fisheries management scenarios examined using LCA for the Tasmanian southern rock lobster fishery. Each scenario represents a CPUE and goal related to different fishery or environment objectives

Fisheries management scenario	**Goal**	**CPUE (kg/trap lift)**	**Reference**
MSY (baseline scenario)	Maximise sustainable catch	0.79	Hartmann et al. (2012)
MSY & no trap limit	Maximise sustainable catch and increase efficiency through removal of an input control	0.79	Hartmann et al. (2012)
MEY	Maximise economic yield	3.8	NZRLIC (2011)
MEY & no trap limit	Maximise economic yield & and increase efficiency through removal of input control	3.8	NZRLIC (2011)
MEY (interim target)	Increase economic yield	1.4	Hartmann et al. (2012)
No-take area	Conservation outcomes through area closure	0.59	Hobday et al. (2005)

Scenario 5 represents an interim harvest target (MEY interim). It is the target for 2020 adopted in the Tasmanian fishery in 2011 following a decision to target MEY as a management objective (Hartmann et al. 2012). It is intended to be a point along a pathway transitioning to higher catch rates closer to long-run MEY, as explored with scenario 3. The MEY interim scenario retains existing input controls in the fishery.

MPAs have been proposed for biodiversity conservation objectives in Tasmania (Marine and Marine Industries Council 2001) and their implementation in the state is therefore plausible. A final scenario, assuming reduced access of the fleet to fishing grounds due to the creation of a network of no-take areas, was based on the adjacent rock lobster fishery in Victoria. The Victorian fishery is the same biological stock as the Tasmanian fishery and therefore affected by the same broad scale trends in recruitment as Tasmania and South Australia on either side (Linnane et al. 2010). MPAs, covering 5.3% of coastal waters, were established in Victoria in 2002 without explicit fisheries objectives (Environment Conservation Council 2000). Prior to the implementation of these parks Hobday et al. (2005) predicted that in the absence of any accompanying reduction to the TAC, the loss of fishable area and biomass available to the industry as a result of the MPA, would reduce catch rates in the area open to fishing. No increase in fish abundance, biomass or egg production across the total of both fished and MPA areas was expected as a result of the MPAs, as it was assumed any increase in biomass inside the MPA would be offset by an equivalent reduction in biomass outside (Hilborn et al. 2006). In Victoria, exploitable biomass and catch rates fell, as anticipated, to a low of 0.59 kg / trap lift following MPA implementation, well below catch rates in South Australia and Tasmania (Hobday et al. 2005).

This experience is drawn on to illustrate the possible effect on CPUE in the TSRL fishery following reduction in resource access of a comparable magnitude, and assuming no other management change such as reduction in TAC or buy-out of commercial quota. In this scenario the management change is therefore restricted to the implementation of MPAs alone and the observed drop in catch rate that occurred in the adjacent Victorian fishery following introduction of MPAs is applied. However, it should be noted that MPAs are often implemented in association with other management changes and that outcomes are strongly influenced by these changes (Yamazaki et al. 2012).

Fuel use intensity (FUI), or litres of fuel required per kilogram of lobster caught, under each management scenario was calculated using the following formula:

$$FUI_X = FUI_{MSY} / (CPUE_X / CPUE_{MSY})$$

Where FUI_X and $CPUE_X$ are the fuel use intensity and CPUE for management scenario x, and FUI_{MSY} and $CPUE_{MSY}$ are the FUI and CPUE for the base case scenario (Table 3.1 and 3.3). Estimated CPUE was also used to prorate the quantity of bait required per kilogram of lobster caught under each scenario.

3.3.4 Data sources for LCA

Fuel use data was derived from fuel expenditure of 20 fishermen. The data was collected through a fleet-wide questionnaire conducted in 2011/12 as part of a larger southern rock lobster project (Econsearch 2012), and converted to litres using historical fuel price data (Motormouth 2012) and the current fuel rebate of 38 cents per litre available to fishers. A standard size boat engine was modelled and sensitivity analysis used to compare results from this model with other boat engines. Catch data was from catch records collected by the Department of Primary Industries, Parks, Water and Environment, Tasmania, as part of the rock lobster quota monitoring system. Materials used in fishing gear, the life of the gear, estimates of bait use and source, and use of antifoulant were calculated based on research conducted at the Institute for Marine and Antarctic Studies (IMAS), University of Tasmania and through interviews with commercial fishers.

Questionnaires on materials and energy used in transport, storage, and packaging were sent to three processors, jointly representing over 50% of rock lobster processors in the state. Packing, or pack out, involves putting lobsters into polystyrene boxes with wood wool and ice packs for export. Data on material, energy and water use were averaged across respondents to create a model of processor resource use and emissions at pack out. The wholesale stage included transport only as TSRL is exported live and requires no further cooling. Transport of lobsters was modelled as airfreight from Tasmania to Beijing via Sydney. Although several other export routes exist, this is the dominant supply route with up to 70% of lobsters airfreighted in 2011. Additional data was sought through the Australasian (Life Cycle Strategies 2012) and Ecoinvent databases (EcoInvent 2012) where it was not available from the sources described above.

Individual LCAs were conducted for the three bait species. Data on Australian salmon was collected from the sole commercial fisher of the Tasmanian fishery (Appendix 2), and data on fuel use for jack mackerel and barracouta was taken from published data (Hilborn and Tellier 2012). Transport and packaging of frozen unprocessed bait fish to Tasmania was modelled using Simapro 7.3.3. Data on freezer storage of bait was provided by processors.

3.4 Results

3.4.1 The environmental impact of exported TSRL– MSY base case

Fishing accounted for a large proportion of the environmental impact of the supply of TSRL across all indicators (Figure 3.2). This stage accounted for 39% of the total global warming potential (GWP) or 12 kg CO_2e, and 188 MJ or 39% of the total cumulative energy demand (CED) (Table 3.2). Fuel use was the main source of the impacts during fishing, accounting for 85% of GWP and 83% of CED. Fishing was also the largest contributor to the eutrophication potential (EP) at 74% or 0.03 kg PO_4e (Figure 3.2), due to the production of nitrogen oxides from the burning of diesel fuel. Fishing was the source of 80% of the total ecotoxicity, largely from antifoul ingredients including copper (oxide and thiocyanate), resins, and zinc oxide.

Airfreight, including domestic and international flights, was the largest contributor to energy use and the carbon footprint, accounting for 55% of GWP, or 17 kg CO_2e kg^{-1} lobster, and 52% of CED, or 253 MJ kg^{-1} lobster. Air transport EP accounted for 24% or 0.01 kg PO_4e kg^{-1} and ecotoxicity from this stage accounted for 18% (Figure 3.2).

Road transport contributed less than 3% of the impacts for each impact category, accounting for less than 1 kg CO_2e kg^{-1} lobster, and 12 MJ kg^{-1} lobster. Inputs for pack out are minimal for live exported SRL and consequently this stage made little contribution to the environmental footprint (Figure 3.2).

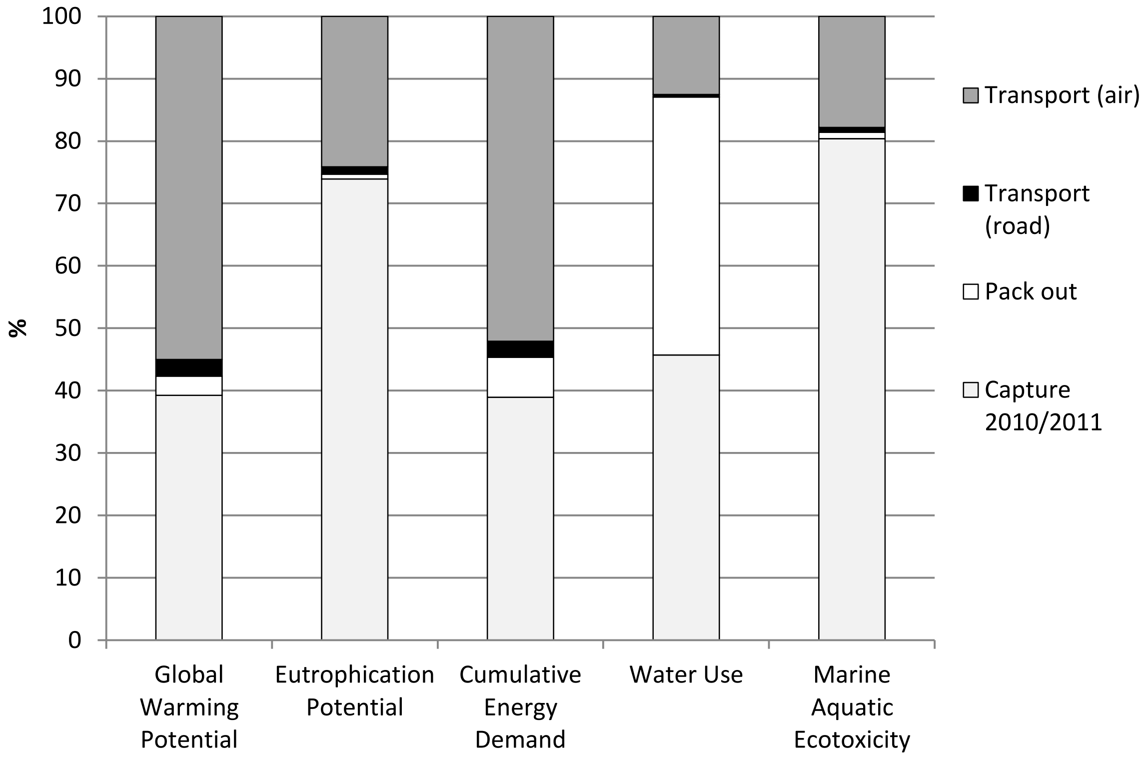

Figure 3.2. Relative proportion of the contribution by impact category to the life cycle impacts of TSRL exported by airfreight from Tasmania to China

Water use was 23L kg^{-1} of live exported lobster. Wash down post pack out accounted for 41%, however, the majority of water used (46%) across the life cycle of TSRL was at the capture stage, mainly for the production of cardboard packaging for transporting bait. Bait use in fishing was the source of 5% of the total global warming emissions and 6% of CED (Table 3.2). Fishing gear (traps) was not a large source of impact in any of the categories measured.

Table 3.2 Life cycle impacts of processes at each stage of lobster production from capture to market

LCA stage	Process	Global warming potential (kg CO_2e)	Eutrophication potential (kg PO_4e)	Cumulative energy demand (MJ)	Water (L)	Marine aquatic ecotoxicity (Day)
a. Capture	Fishing boat engine	11	2.54E-02	157	1.2	4.72E-11
	Bait	1.5	2.00E-03	30	5.6	1.03E-11
	Trap	0.1	2.11E-04	1.6	3.3	8.12E-11
	antifoul	0.01	1.20E-04	0.17	0.42	2.50E-10
Sub total		**13**	**0.03**	**189**	**11**	**3.89E-10**
b. Pack out	Electricity	0.9	1.51E-04	24.00	1.10	3.04E-12
	Water	8.14E-04	2.06E-07	0.02	7.90	5.79E-15
	Wood wool	9.97E-03	2.81E-05	3.56	0.30	1.37E-12
	Poly box	5.07E-02	9.49E-05	3.53	0.15	0.00
	Gel pack	3.56E-03	1.02E-05	0.14	0.06	3.73E-13
Sub total		**0.96**	**2.84E-04**	**31**	**9.5**	**4.79E-12**
c. Transport	Road	0.85	4.35E-04	12	0.09	3.72E-12
	Domestic airfreight	2	1.07E-03	30	0.33	1.01E-11
	International airfreight	15	8.06E-03	223	2.6	7.61E-11
Sub total		**18**	**9.57E-03**	**265**	**3**	**8.99E-11**
Total		**31**	**0.04**	**485**	**23**	**4.83E-10**

3.4.2 Unintended consequences of fisheries management decisions

Capture stage

The environmental impacts at the capture phase of TSRL modelled through the scenarios were substantially affected by the fisheries management decisions examined (Figure 3.3, Table 3.4). Changing the fisheries management harvest target from maximising the amount of catch (MSY) to maximising economic yield (MEY) reduced the footprint of the fishery across all indicators and improved fuel use efficiency by 2.7 L kg^{-1} (Table 3.3). Management for MEY reduced GWP by 10 kg CO_2e kg^{-1} lobster or 80%, thereby reducing the relative importance of the capture stage in the environmental footprint from 42% to 12% GWP (Figure 3.3). Changing the management focus to MEY improved CED by 152 MJ kg^{-1} lobster or 80% at capture (Table 3.4).

Table 3.3 Fuel use intensity and standard deviation of a range of management scenarios examined in the Tasmanian southern rock lobster fishery

Management scenario	MSY	No trap limit	MEY	MEY & no trap limit	MEY interim	No-take area
Fuel use intensity (l kg^{-1} lobster ± SD)	3.3 ± 1.7	1.7 ± 1.7	0.63 ± 0.25	0.31 ± 0.25	1.7 ± 0.68	4.3 ± 1.9

Removing the trap limit in the fishery, when targeting MSY, reduced the magnitude of the environmental impacts for all indicators at capture. The carbon footprint decreased by 5.4 kg CO_2e kg^{-1} lobster or 44%, EP decreased by 46%, and CED reduced by 79 MJ kg^{-1} lobster or 42%. Ecotoxicity also decreased by 6% (Table 3.4). Removing the trap limit in the fishery when managed for MEY resulted in the greatest reductions across all indicators, with emissions reduced from 13 kg CO_2e kg^{-1} lobster to 1.4 kg CO_2e kg^{-1} lobster, a reduction of 89%, or 11.6 kg CO_2e from the base case scenario of MSY with no reduction in input controls.

Targeting MEY and removing trap limits reduced EP, due to decreases in fuel use and reduction in NOx emissions. The EP was reduced by 80%, or 0.02 kg PO_4e kg^{-1}, under the MEY management scenario and by 89% under the MEY and no trap limit scenario. Similarly, ecotoxicity was reduced by 13% from the MSY base case under the MEY and no trap limit scenario (Table 3.4).

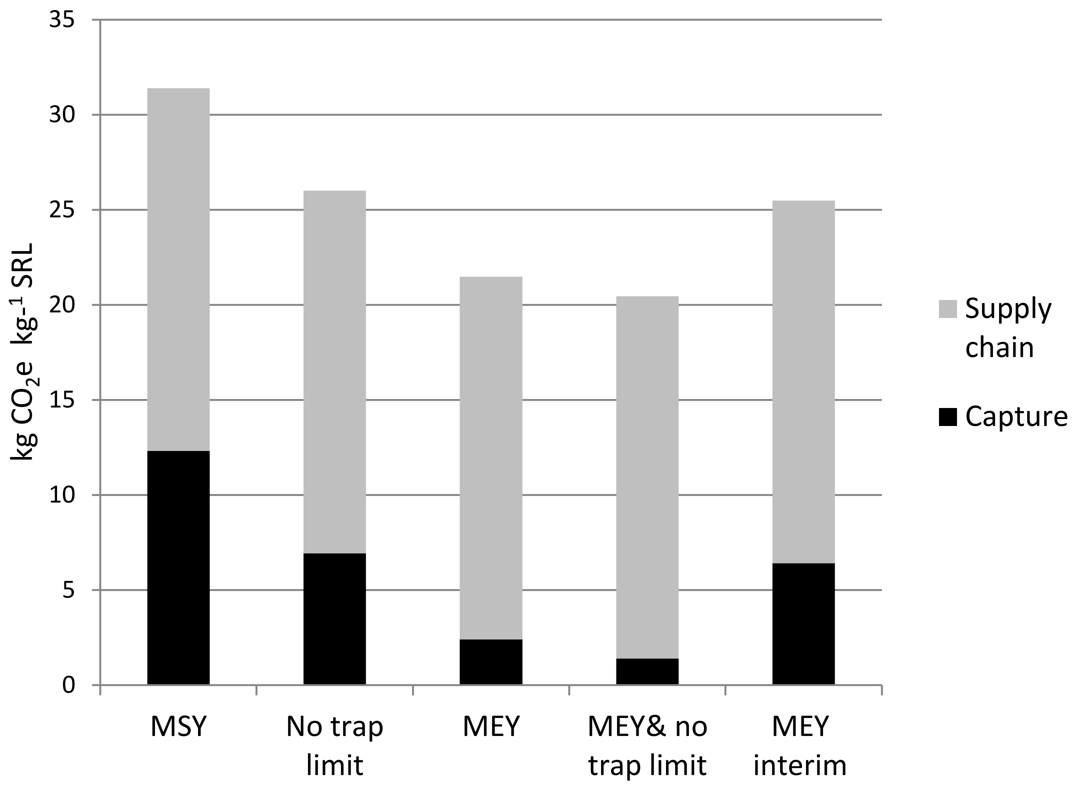

Figure 3.3 Impact of different fishery management scenarios on global warming potential, kilograms of carbon dioxide produced per kilogram of TSRL

Reductions across all indicators were observed under the MEY interim management scenario. GWP and CED were reduced by 6 kg CO_2e kg^{-1} or 48% and 90 MJ kg^{-1} or 48% from the base case (Table 3.4). EP, ecotoxicity and water use were also reduced by 48%, 7% and 28% respectively.

Table 3.4 Percentage change in indicators relative to MSY (base case) under different management scenarios

Management scenario		No trap limit	MEY	MEY & no trap limit	MEY interim	No Take area
Global Warming Potential % Change	Capture	-44	-80	-89	-48	23
	Supply chain	-17	-32	-35	-19	8.8
Eutrophication Potential % Change	Capture	-46	-80	-89	-48	22
	Supply chain	-34	-59	-66	-35	16
Cumulative Energy Demand % Change	Capture	-42	-80	-88	-48	23
	Supply chain	-16	-31	-34	-19	8.9
Marine Aquatic Ecotoxicity % Change	Capture	-6	-12	-13	-6.9	.6
	Supply chain	-5.6	-11	-12	-6.6	3.4
Fresh water use % Change	Capture	-5.5	-51	-52	-29	20
	Supply chain	-2.5	-23	-24	-13	9

The introduction of no-take MPAs increased the environmental impacts of TSRL supply across all indicators (Table 3.4). The GWP of the fishery increased by 23% or 2.8 kg CO_2e kg^{-1} lobster and the CED increased by 23% or 43 MJ kg^{-1} lobster (Table 3.4). The reduction in access to the resource, without corresponding adjustment in the TAC of the fishery, increased the EP by 22% or 0.01 PO_4e kg^{-1} lobster (Table 3.4) and ecotoxicity by 3.6%.

Fresh water use was affected by changes in fishery management. Targeting MEY with no trap limit reduced water use by 52%. Water use in the no-take MPA scenario, in contrast, increased by 2.1 L kg^{-1} or 20%.

Supply Chain

Given the contribution of airfreight to the environmental footprint of the SRL, the percentage change in indicators relative to MSY (base case) under different fishery management scenarios is lower when the whole supply chain is considered (Table 3.4, Figure 3.3). Changing the fisheries management target to MEY reduced CO_2e emission by 32% and improved CED by 31% across the supply chain. Removing the trap limit in the fishery, when targeting MSY, decreased the carbon footprint by 17% and also decreased CED, EP, water use and ecotoxicity (Table 3.4). A change in management to target MEY with no input control decreased the overall carbon emissions by 35% and reduced water use by 24% across the supply chain. This reduction equated to 5.5 L kg^{-1} live exported lobster.

Adopting a transitional, or interim, MEY harvest target reduced CO_2 emission across the supply chain by 19% or 6 kg CO_2e kg^{-1} lobster from the base case (Table 3.4). This reduction occurred under the interim MEY where the CPUE (1.4 kg / trap lift) was less than double the base case (0.79 kg / trap lift). Managing the fishery for MEY increased CPUE to 3.8 kg / trap lift, nearly a fivefold increase, resulting in a reduction of 10 CO_2e kg^{-1} of lobster across the supply chain.

The introduction of no-take MPAs increased the environmental impacts of SRL supply across all indicators (Table 3.4). The reduction in access to the resource, without any adjustment in management in the fishery, resulted in an increased GWP of nearly 9% in the model (Table 3.4).

3.5 Discussion

This study demonstrates the application of the life cycle approach to a trap fishery and shows that the outcomes of the scenarios are sensitive to management and stock status at the time of assessment. Under the outlined assumptions, simply altering trap limits had a profound effect on all LCA indicators examined and this type of decision is made regularly in fisheries management. Many seafood LCAs provide a snapshot view of impacts that relate to stock status and management rules at a particular point in time (see for e.g. Ziegler and Valentinsson 2008; Vázquez-Rowe et al. 2011) . The sensitivity of LCA results to stock status is important because the results of seafood LCAs can be compared to assess the relative impacts of different fisheries (Hospido and Tyedmers 2005), seafood products (Hall et al. 2011) or other food products (Ellingsen and Aanondsen 2006; Nijdam et al. 2012). Understanding not only the level of sensitivity, but also the source, will provide more useful comparisons of the relative impacts of different products.

3.5.1 Life cycle impacts of the southern rock lobster

Under current stock levels, which resulted from management targeting higher catch levels, the TSRL has been independently assessed as a sustainable fishery (FRDC 2012). However, fishing is a major contributor to a range of environmental impacts that are not assessed or managed under current fishery management practices. The TSRL fishery is fuel intensive, more so than many other fisheries, including purse seine caught pelagic fish which have a reported FUI between 21 and 360 L t^{-1}, demersal trawl fisheries with FUI between 530 and 2447 L t^{-1}, as well as other lobster trap and trawl fisheries (Table 3.5). The high FUI results in a higher contribution to global warming potential (GWP) per kilogram. FUI is typically much higher in high value, low volume fisheries such as lobster fisheries than for many other fishing techniques (Tyedmers and Parker 2012), a reflection of high prices allowing the fishery to remain profitable despite high input costs.

Table 3.5 Reported fuel use intensities for lobster fisheries

Target lobster	**Species name**	**Year**	**Gear**	**FUI (l/t)**
European	*Nephrops norvegicus*	2009	Trap	306
American (Maine)	*Homarus americanus*	2011	Trap	991
American (Nova Scotia)	*Homarus americanus*	2011	Trap	1026
Norway	*Nephrops norvegicus*	2001	Trawl	1030
Norway	*Nephrops norvegicus*	2004		1160
Norway	*Nephrops norvegicus*	2009		1224
Norway	*Nephrops norvegicus*	2008	Creel	2156
Australian (South Australia)	*Jasus edwardsii*	2011	Trap	2975[a,b]
Australian (Tasmania)	*Jasus edwardsii*	2011	Trap	3330[c]
Norway	*Nephrops norvegicus*	2008	Trawl	4119
New Zealand	*Jasus edwardsii*	2012	Trap	4731[d]

Adapted from Tyedmers and Parker (2012), [a,b] (Econsearch 2008b, a), [c] this chapter [d] (Hilborn and Tellier 2012)

The capture phase typically accounts for the greatest share of total greenhouse gas emissions in developed country fisheries (Thrane 2004a, 2006; Ziegler et al. 2003; Ziegler and Valentinsson 2008; Hospido and Tyedmers 2005; Parker 2011; Vázquez-Rowe et al. 2013). In addition to high fuel intensity at capture, around 70% of TSRL are airfreighted live from Australia to China. Airfreight contributed more to the carbon footprint and energy use of the SRL than the capture stage. The results are not surprising given that air transport is the least efficient method of transport in terms of emissions (Facanha and Horvath 2006) and has the highest environmental impact per tonne of any mode of transport (Smith 2005). High carbon emissions from airfreight have also been measured for aquaculture fish transported from Norway to East-Asia and the United States of America, (Andersen 2002) and for fresh and frozen salmon from Norway to Japan (Winther et al. 2009).

The TSRL is considered a luxury good and demand from emerging economies for these products is predicted to grow, with China to account for over 20% of global luxury sales by 2015 (Atsmon et al. 2011). Asian consumers place a price premium on live fish and the resulting high prices means that production (and emissions) are less sensitive to change in fuel price and can be expected to grow with demand. At present, fish, crustaceans, molluscs and preparations account for 8.8% of air freight exports from Australia by weight (Hamal 2011) however this will likely increase given the predictions of demand growth for live seafood.

The contribution of bait to the environmental footprint of the TSRL at the capture stage was approximately 12% of the global warming emissions. Bait also contributed 16% of the cumulative energy demand (CED) kg^{-1} lobster, or 1833 tonnes CO_2e per year for the Tasmanian fishery. These figures as a proportion of total emissions are higher than a comparable assessment of the Norway lobster (*Nephrops norvegicus*). In that fishery, where baited wooden traps are set in links on the seafloor, bait was the source of 10% of the total energy consumption and 5% of global warming emissions (Ziegler and Valentinsson 2008). The rate of bait use in the TSRL fishery in 2010/11 was approximately 1 kg of bait for 1 kg of lobster, and while this figure seems high, inputs of bait often exceed product outputs in capture fisheries (Ayer et al. 2009). Harnish and Willison (2009), for example, estimate a ratio of 1.9:1 bait to catch for the Nova Scotia lobster fishery. The majority of bait used in the TSRL fishery is sourced from Australia and New Zealand, however, suppliers have indicated they are increasingly sourcing bait from India and South Africa.

The use of antifouling agents reduces growth of marine organisms, which if left on the boat hull can seriously reduce fuel efficiency and increase fuel use (Evans et al. 2000) and subsequent greenhouse gas emissions (GHGe). However, their use comes with well documented ecotoxicity costs (Fernández-Alba et al. 2002). For example, the use of antifoul has been identified as contributing disproportionately to the total environmental impact of Spanish tuna fisheries (Hospido and Tyedmers 2005). There is some debate over the importance of the toxicity of metals used in antifoul with an argument that a zero toxicity characterisation factor should be applied in marine waters as the oceans are deficient in essential metals, such as zinc and copper (Aboussouan et al. 2004).

Water use is often not included as an indicator in LCAs, in particular for wild fisheries. Water use in the supply of TSRL was 23 L kg^{-1} live lobster, which equates to approximately 71 L kg^{-1} of lobster meat, with a 30% recovery rate, (as used by Hilborn and Tellier 2012; van Putten et al. 2015). To put this in context, red meat production in Australia has been calculated by different authors at 18–540 L kg^{-1} beef and sheep meat (Peters et al. 2010) and 17,112 L kg^{-1} beef (Hoekstra and Chapagain 2007), while the global average water use for beef production has been calculated at 15,415 L kg^{-1} (Mekonnen and Hoekstra 2012). Water use for the TSRL is therefore comparatively low and when calculated for the entire annual catch, amounts to 28 ML per year, equivalent to the annual per capita household consumption of 320 people in Australia (ABS 2012). Seafood may, therefore, present a good option when selecting food on the basis of water use, an important consideration given that water availability is likely to limit future food production (Hanjra and Qureshi 2010).

3.5.2 Effects of management decisions

Modelled management changes in the TSRL fishery to target MEY lead to outcomes such as improved energy efficiency and lowered carbon emissions at capture, in turn reducing the overall GWP, CED and EP of the TSRL. Improvements in these indicators would be achieved with any move in the direction of MEY such as the interim target currently in place in Tasmania. Improvements in the environmental performance of the fishery can also be achieved by relaxing the input control of trap limits. This limit on the number of traps that can be used per vessel was originally established to control total effort; however it now serves mainly to limit contraction of the fleet and to spread effort. Analysis shows that this management strategy has the unfortunate outcome of increasing emissions. Quantifying the effect of trap limit regulation on emissions will help promote discussion of other management tools that may produce the same fishery outcome with less environmental impact.

Fisheries management decisions can therefore strongly influence the overall environmental footprint of seafood products including energy use and resulting GHGe. The same observation was made of the New England Atlantic herring fishing where changes to trawl input controls and the TAC would reduce fuel use and emissions (Driscoll and Tyedmers 2010). Impacts on ecosystems are more complicated, as illustrated by the Swedish Nephrops trawl fisheries, where grid trawls reduced bycatch mortality of cod but also reduced environmental performance by the emmission criteria used in this study (Hornborg et al. 2012).

In the MPA scenario, the closure of parts of the fishery resulted in the TAC being harvested from a smaller area so that exploitation rates increased and abundance declined (Hobday et al. 2005). The unintended cost of MPAs was a greater environmental footprint due to reduced fuel use efficiency and increased bait use per kilogram of lobster. This outcome highlights the need for marine resource managers to ensure that catch and effort displaced by MPAs is removed from the fishery (McGarvey and Linnane 2009).

3.6 Conclusions

The life cycle perspective holds considerable promise for informing seafood environmental policy (Pelletier et al. 2007). The consideration of environmental metrics provided through LCA further complicates an already challenging task involving often competing fisheries management objectives and ethical choices such as between employment, economic yield and providing food (Béné et al. 2010; FAO 2012). Nonetheless the importance of resource use and emissions reduction suggests that fisheries management needs to consider a wider range of environmental impacts, in particular GWP and CED. As was found with TSRL, there are likely to be opportunities for revising fisheries, and more generally marine resource, management to give improved outcomes for both traditional sustainability measures and also a broader suite of environmental impacts.

CHAPTER 4: DOMESTIC OR IMPORTED? AN ASSESSMENT OF CARBON FOOTPRINTS AND SUSTAINABILITY OF SEAFOOD CONSUMED IN AUSTRALIA

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4.1 Abstract

The distance between where food is produced and consumed is increasing, and is often taken as evidence of an unsustainable global food system. Seafood is a highly traded commodity yet seafood sustainability assessments do not typically consider the impacts of the movement of products beyond the fishery or farm. Life cycle assessment is used to examine the carbon footprint of the production and distribution of select seafood products that are consumed in Australia and determine differences in the sustainability of imports and their domestically produced counterparts. The distance food is transported was found not to be the main determinant of food sustainability. Despite the increased distance between production and consumption, carbon footprints of meals from imported seafood are similar to meals consisting of domestically produced seafood, and sometimes lower, depending on the seafood consumed. In combining LCA with existing seafood sustainability criteria the trade-offs between sustainability targets become more apparent. Carbon 'footprinting' is one metric that can be incorporated in assessments of sustainability, thereby demonstrating a broader perspective of the environmental cost of food production and consumption.

4.2 Introduction

Chapter 3 highlighted the substantial contribution of the export stage to the overall product footprint and the importance of impacts occurring as a result of the movement of seafood beyond the fishery, and beyond national borders. In this chapter we examine the transport of seafood products in more detail. Global food trade is increasing at a faster rate than food production (Ercsey-

Ravasz et al. 2012) and population growth (FAOSTAT/Tradestat 2009) and the distances between production and consumption are rapidly increasing (Thomas et al. 2014; Watson et al. 2015a). Global supply chains place great demands on ecosystems and natural resources (Wible et al. 2014; Tilman and Clark 2014) and localised food systems have been promoted within academic literature, public policy and alternative food movements as a more sustainable option (La Trobe and Acott 2000; Hendrickson et al. 2002; Legislative Assembly of Ontario 2013; Lang and Heasman 2009). Trends in trade of fish and fishery products run counter to aspirations of localised production as they are some of the most-traded food commodities worldwide (FAO 2014b), with the world's major importers, the United States of America (USA) and Japan dependent on imports for about 60% and 54%, respectively, of their seafood consumption (FAO 2012).

Compared to agriculture, fisheries are poorly represented in food policy (Lang and Heasman 2009) and sustainable seafood policies are being developed in isolation from other food policy. Conventionally, seafood sustainability has tended to be focused on issues concerning the harvesting of fish as a natural resource (Olson et al. 2014) and as a result, management of sustainability within capture fisheries is concerned with ecological issues such as overfishing, stock biomass and recruitment, and in some more complex management regimes, ecosystem impacts and bycatch through an ecosystem-based fishery management (EBFM) approach (Zhou et al. 2010). Similarly, management of sustainability in aquaculture systems is largely concerned with production issues including impacts of invasive species on local biodiversity (Silva et al. 2009), disease control (Bondad-Reantaso et al. 2005), impacts of chemical use on environmental and human health (Burridge et al. 2010), eutrophication of natural waterways, sensitive land conversion, and the use of wild fish in feed (Cao et al. 2015; Naylor et al. 2000; Diana 2009). Consideration of the broader supply chain impacts of seafood supply is relatively recent (Avadí and Fréon 2013; Parker 2012; Henriksson et al. 2012b).

Rising GHGe are affecting food production from the land and sea (IPCC 2014; Campbell 2014) and the supply of seafood contributes to these rising emissions (Tyedmers et al. 2005). Achieving a more holistic determination of seafood sustainability requires consideration of emissions generated along seafood supply chains, such as product carbon footprints, as well as traditional measures of sustainability at capture or culture. The issue of human equity is also inextricably linked to environmental quality and measures of social justice, equity, rights and people's quality of life should also be considered in assessments of sustainability (Agyeman 2008). Australia provides an interesting case study for examining different sustainability measures, and the compatibilities or trade-offs that emerge between them. Australia has been ranked in the top five countries for fisheries management (Pitcher et al. 2009b) and the majority of commercial fish stocks in Australia have been assessed as sustainable (Woodhams et al. 2013). However, nearly 72% of the seafood consumed in Australia is imported (Ruello 2011) and growth in consumption of imports is expected to continue into the future, in line with government food frameworks (DAFF 2013) and to meet consumer demand for low-cost seafood products (Department of Agriculture 2013).

This paper quantifies an aspect of sustainability that is not typically assessed in the production and distribution of select seafood products available in Australia, the carbon footprint (CF). Life cycle assessment (LCA) is used to compare the CF of three domestic wild-capture products with imports that are readily substituted by consumers. Patterns in the emissions of different species, production methods and supply chain stages are identified, and these results are examined in the context of existing seafood sustainability assessments. The trade-offs and opportunities in combining LCA with existing seafood sustainability criteria are also identified and the need for broader assessments to operationalise holistic, system-wide concepts of food sustainability and inform emerging food policy are discussed, in particular in terms of reducing carbon emissions.

4.3 Methods

4.3.1 Australian seafood imports

Australia's seafood imports consist mainly of lower-value products such as frozen fillets, frozen prawns (where 'prawns' refers to both shrimp and prawn within Caridea and Dendrobranchiata) and canned fish (Department of Agriculture 2013). Frozen and thawed catfish (*Pangasius)* fillets from farms in Vietnam are now the most commonly eaten import (Ruello 2011). A small amount of high value products such as lobster and abalone are also imported. The four most important sources of seafood imports to Australia are Thailand, New Zealand, Vietnam and China (Ruello 2011), however, prawn, fish and lobster imports are sourced from around 100 different countries (ABARES 2012).

Most seafood imported into Australia is sent by ship with approximately 10% sent by airfreight. Almost all annual imports of prepared or preserved prawns are sent by sea (ABARES 2012). In contrast, some products such as fresh or chilled fish fillets (Australian Customs Service statistical code 304100042) are mostly airfreighted. The majority of lobster imported into Australia is frozen and transported by sea. Small volumes of fresh lobsters are flown to Australia from South East Asia and New Zealand, some of which are re-imports which have been caught in Australia and sent overseas for processing.

Data purchased from the Australian Bureau of Statistics (ABS) was used to calculate volume, country of origin and transport mode for several product categories of imported prawn, fish and lobster over the past 10 years (www.abs.gov.au). While no data was available for some product groups, 82% of imports were included in this study.

4.3.2 Life cycle assessment

LCA is an integrated tool for quantifying and comparing potential environmental impacts throughout the life cycle of a product or products. The methods used in LCA are standardised through the International Organization for Standardization (ISO 2006b). In this study results from LCAs on four select Australian fisheries are compared: Tasmanian southern rock lobster (*Jasus edwardsii*), white banana prawn (*Fenneropenaeus merguiensis*) from the Northern Prawn Fishery, Australian salmon (*Arripis trutta)* fished in Tasmania and flathead [3] (*Neoplatycephalus richardsoni*) from the Commonwealth Trawl Fishery (CTS), with products included on the Australian Bureau of Statistics (ABS) list of imports, documented in six peer-reviewed LCAs, one conference paper and two PhD theses (see Appendix 3, Tables A3.1, A3.4 and A3.5). These studies cover three of the five most consumed seafood groups in Australia, including prawns, fish consumed crumbed/battered - predominantly imported catfish (Ruello 2011)- and Atlantic salmon (Danenberg et al. 2012), as well as a luxury seafood and several less popular fish species.

The LCA was modelled using SimaPro Software version 7.1.6. with the impact assessment method CML-IA baseline, developed by the Center of Environmental Science (CML) of Leiden University (Universiteit Leiden 2015). All studies included use the same data libraries and LCA impact assessment method as recommended by (Baumann and Tillman 2004). To ensure maximum comparability between studies on one impact category, the global warming potential (GWP), was selected as the focus (Henriksson et al. 2015), and was based on the characterisation model

[3] We use flathead to represent the CF of catch from the CTS, of which flathead is a key species

developed by the Intergovernmental Panel on Climate Change (IPCC), where the GWP for a time horizon of 100 years (GWP100) is expressed in kilograms of carbon dioxide equivalent. The CF was assumed to be equivalent to GWP, where both are measured in units of CO_2e. The functional unit (FU) for all products is 1 kg of whole product. Where transport is included, the FU is 1 kg frozen product when transported by sea. For canned Atlantic salmon, the FU is 1 kg whole fish and the transport method is seafreight, but energy use for the refrigerated container is not included. For wild-capture Australian prawns the FU is whole frozen product and for southern rock lobster the FU is live product.

All studies employed mass allocation. Published LCAs using other allocation methods were excluded from analysis. Original data was collected for the Australian LCAs and sourced from Ecoinvent libraries where not otherwise available. The system boundary for all wild-capture studies included fuel, gear and bait up to the point of landing but excluded infrastructure. Sensitivity analysis was performed where variation existed between studies regarding the inclusion of refrigeration and refrigerants on boats. Aquaculture studies included feed and energy use up to the point of harvest, except for the salmon study which included feed only (Pelletier and Tyedmers 2007). For transport of imports by boat to Australia the fuel use for the journey and the refrigerated container for frozen products were included. Harbour activities have not been included. Fuel use was modelled for airfreight for the journey.

Sensitivity analysis is also performed on the aspects considered to have the greatest impact on overall results: feed conversion ratio (FCR) and catch per unit effort (CPUE). The effect of lowering or raising the FCR on the CF was modelled for aquaculture species, assuming other factors remain the same including feed composition, and energy use and emissions associated with feed production. For wild-capture species the impact on results of changes in fuel use over time as a result of changing CPUE was modelled. The following assumptions were made: 1. catch rate scales effort and therefore fuel use; 2. all emissions are perfectly variable with catch rates and there are no fixed emissions, i.e. the fleet will rescale with catch rate, for example, if catch rate doubles then fuel emissions halve, because trips, bait, and gear required halve; 3. the fleet is homogenous so when the fleet rescales with catch rate an average vessel enters or leaves the fishery and the composition of the fleet, in terms of efficiency of individual vessels, stays the same.

4.4 Results

4.4.1 Carbon footprint of seafood in Australia

The capture or farm stage was typically the major source of carbon emissions for frozen seafood transported by sea. Carbon emissions from seafreight were less than 1 kg CO_2e kg^{-1} seafood (Figure 4.1). For prawns, these emissions accounted for 4% of the CF for transport from China and the Philippines to Australia (Figure 4.1a). Trap-caught *Homarus americanus* landed in the USA and shipped to Australia had a smaller CF at wholesale in Sydney than did the Australian southern rock lobster both at landing in Tasmania and at wholesale in Sydney (Figure 4.1b). It is notable that the CF increased by over 400% or 18 kg CO_2e kg^{-1} when *H. americanus* was flown (Boston - Los Angeles – Sydney, main flight path), instead of shipped (Boston - Middle East – Sydney, main sea route) to Australia from the USA. Only small amounts of lobster are currently flown to Australia from the USA, all of which are frozen.

Emissions from seafreight accounted for less than 10% of total emissions for catfish from Vietnam, canned salmon from the USA, and hake from Spain, while they accounted for over 60% for sardines from Portugal (Table 4.1). For fish species with low emissions at the production stage, a modest increase in total carbon emissions from seafreight resulted in substantial percentage increases in the CF. The addition of seafreight to 1 kg of sardines from Portugal, for example, resulted in a 157% increase of the CF despite only increasing emissions per kilogram sardine by approximately half a kilogram CO_2e.

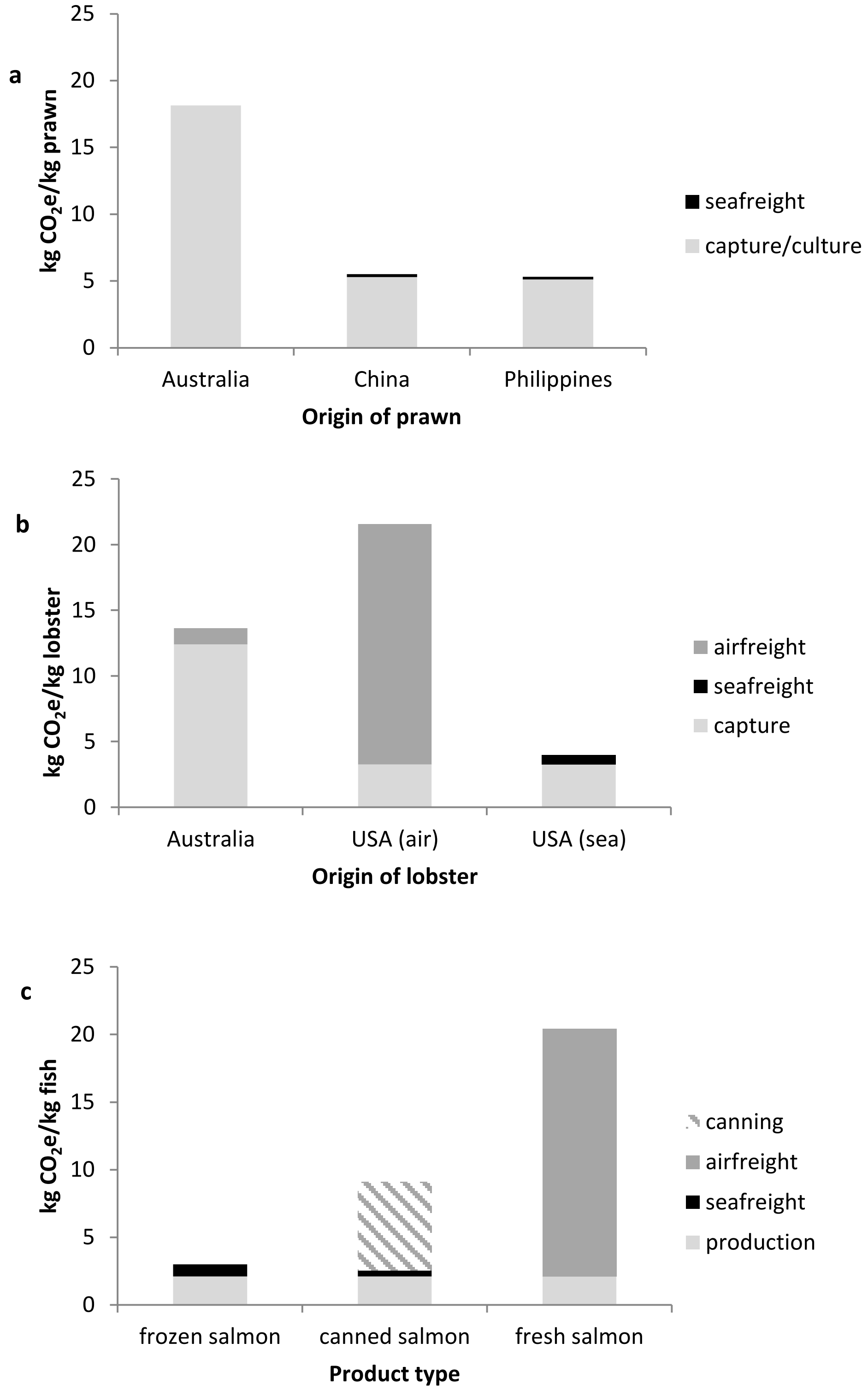


Figure 4.1. Carbon footprint of 1 kg whole seafood with supply chain stages
a. whole frozen prawn with refrigerated seafreight to Australia; b. whole frozen lobster: *H. americanus* with sea- and airfreight to Sydney, and live *J. edwardsii*; c. Whole Atlantic salmon with canning and transport - frozen salmon includes refrigerated transport, fresh salmon does not include refrigeration. See Figure 4.2 for sensitivity of results by species.

Table 4.1 Carbon emissions for different fish products at production, processing and transport

Fish	**Origin**	**Production** $CO_2e\ kg^{-1}$	**Sea freight** $CO_2e\ kg^{-1}$	**Air freight** $CO_2e\ kg^{-1}$	**Canning** $CO_2e\ kg^{-1}$	**Total** $CO_2e\ kg^{-1}$	**Production % total**	**Transport % total**	**Canning % total**
Catfish (*Pangasianodon hypophthalmus*)[1]	Vietnam	8.9	0.2			**9.1**	98	2	
Catfish (*P. hypophthalmus*)[1]	Vietnam	8.9		7.7		**17**	54	46	
Hake (*Merluccius merluccius)*[2]	Spain	4.8	0.53			**5.3**	90	10	
Flathead (*Neoplatycephalus richardsoni)*[3]	Australia	2.4				**2.4**	100		
Frozen salmon (*Salmo salar)*[4]	USA	2.1	0.7			**2.8**	75	25	
Canned salmon (*S. salar)*[4]	USA	2.1	0.3		6.6	**9**	24	3	73
Fresh salmon (*S. salar)*[4]	USA	2.1		18.3		**20**	10	90	
Horse mackerel (*Trachurus trachurus*)[2]	Spain	1.85	0.53			**2.6**	72	28	
Australian Salmon (*Arripis trutta*)[5]	Australia	0.97				**1**	100		
Sardine (*Sardina pilchardus*)[2]	Spain	0.74	0.53			**1.3**	58	42	
Sardine (*S. pilchardus)*[6]	Portugal	0.36	0.56			**0.9**	38.9	61	

[1] Bosma et al., 2011, [2] Iribarren et al., 2010, [3] Appendix 1, [4] Pelletier and Tyedmers, 2007, [5] Appendix 2, [6]Almeida et al., 2014. Sensitivity of results by species presented in Figure 4.2

Transport of frozen salmon from the USA to Australia resulted in emissions of 0.7 kg CO_2e kg^{-1}, which accounted for 25% of the CF (Figure 4.1c). The transport stage of canned salmon, which does not require refrigeration, accounted for 0.3 kg CO_2e kg^{-1}, while the canning process was responsible for 6.6 kg CO_2e kg^{-1} or 73% of carbon emissions (Figure 4.1c). The farming of salmon accounted for 32% of the CF for canned salmon and 82% for frozen salmon. For airfreighted salmon from the USA, carbon emissions increased by 18 kg CO_2e kg^{-1} relative to seafreight (Figure 4.1c). Airfreight accounted for 57% of the CF for catfish from Vietnam and one kilogram of airfreighted catfish had a CF 12 kg CO_2e larger than if sent by sea. Seafreight of catfish, in contrast, accounted for only 2% of carbon emissions and resulted in 0.2 kg CO_2e kg^{-1} (Table 4.1).

4.4.2 Comparison of carbon footprint at landing or harvest by species

Carbon emissions varied between different species of fished and farmed seafood. Wild-caught *Penaeus esculentus*, an endemic Australian prawn, had the highest CF of all the seafood examined in this study (see Appendix 3 for full list of species), accounting for 32 kg CO_2e kg^{-1} (Chapter 2, Farmery et al. 2015) (Figure 4.2a). Farmed *P. monodon* prawns had lower emissions at 5.1 kg CO_2e kg^{-1} (Baruthio et al. 2008), similar to that of farmed *Litopenaeus vannamei,* 3.1 kg CO_2e kg^{-1} (Cao et al. 2011). Emissions related to wild-caught banana prawns (*Fenneropenaeus merguiensis)* were similar to that of farmed prawn species, at 4.2 kg CO_2e kg^{-1} (Chapter 2, Farmery et al. 2015). *Jasus edwardsii* lobsters from Australia, had higher carbon emissions, 12.3 kg CO_2e kg^{-1} (Chapter 3, Farmery et al. 2014), than *Homarus americanus,* 4.4 kg CO_2e kg^{-1} (average of USA and Canada) (Driscoll 2008; Boyd 2008) (Figure 4.2b).

Catfish *(Pangasianodon hypophthalmus)* had the highest CF of all fish species at 9 kg CO_2e kg^{-1} (Bosma et al. 2011) (Figure 4.2c). Hake (*Merluccius merluccius*) had a lower footprint than catfish, 5.3 kg CO_2e kg^{-1} (Iribarren et al. 2010b) but a higher footprint than frozen salmon (*Salmo salar*) and flathead (*Neoplatycephalus richardsoni*) which both had emissions around 3 kg CO_2e kg^{-1} (Pelletier and Tyedmers 2007, Appendix 1). Horse mackerel (*Trachurus trachurus)* had emissions of 2.4 kg CO_2e kg^{-1} (Iribarren et al. 2010b) while sardines (*Sardina pilchardus)* (Almeida et al. 2014; Vazquez-Rowe et al. 2014) and Australian salmon (*Arripis trutta*) (Appendix 2) had a CF of 1 kg CO_2e kg^{-1} or less, the smallest of all seafood examined (see section 3.3 for sensitivity of results).

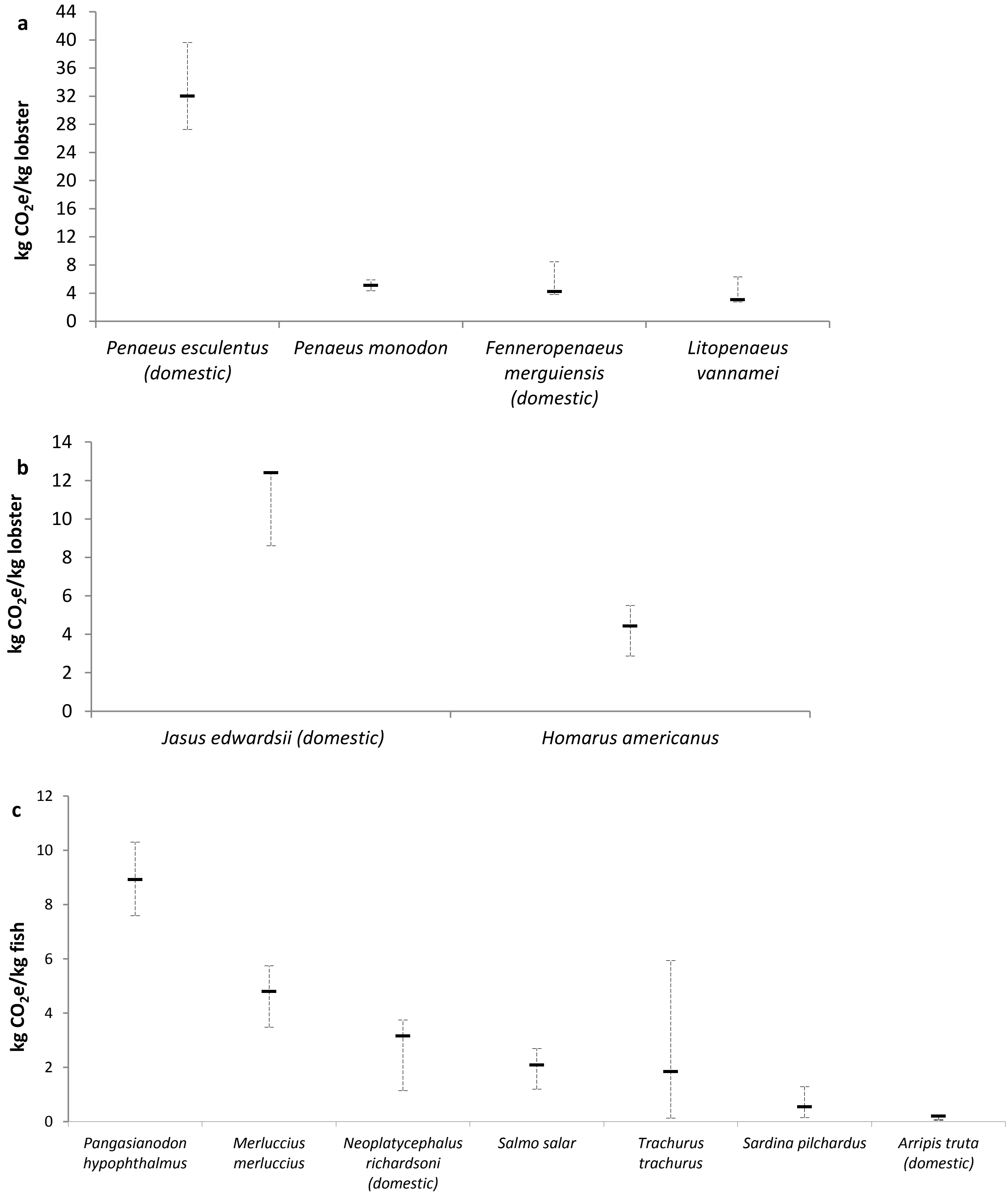
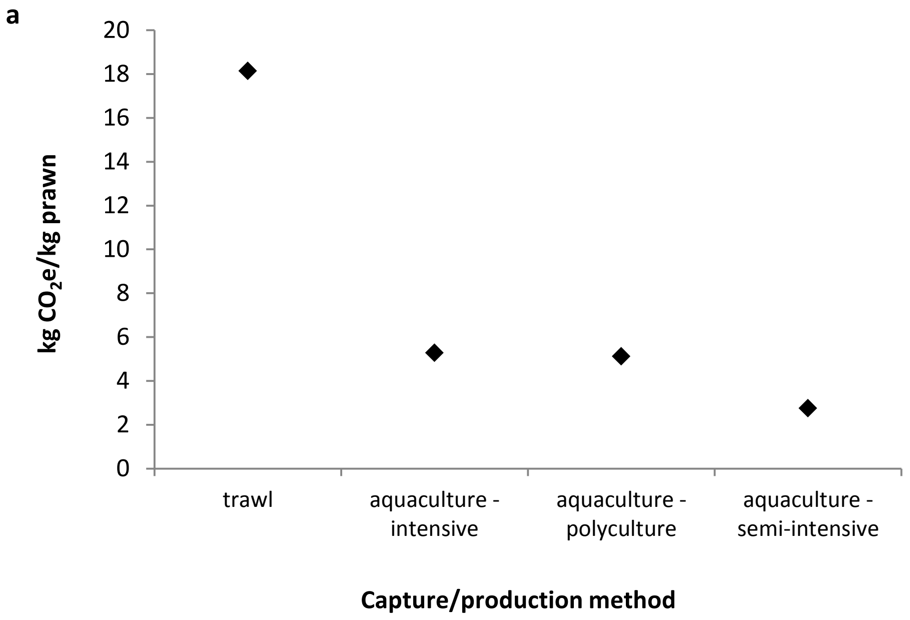


Figure 4.2 Carbon footprint of 1 kg whole prawn, whole lobster and whole fish at landing/ farm gate. Error bars represent range of reported results based on variation in catch per unit effort over time for wild-capture and different feed conversion ratios for aquaculture.

4.4.3 Comparison of carbon footprint at landing and harvest by production method

Three different types of prawn aquaculture had lower CF than the Australian trawl caught prawns: polyculture (Baruthio et al. 2008), intensive, and semi-intensive (Cao et al. 2011) (Figure 4.3a). Emissions from the two prawn trawl fisheries were averaged at 18 kg CO_2e kg^{-1} (Chapter 2, Farmery et al. 2015), however, the CF of trawling for banana prawns was similar to that of aquaculture prawns. All lobsters were trap caught therefore no comparison between methods was made.

There was substantial variation between fishing methods for finfish reported in the literature, although studies on different methods were not available for all species. Flathead caught by otter-trawl in Australia had a footprint more than double those caught by Danish seine, 3.5 kg CO_2e kg^{-1} compared with 1.3 kg CO_2e kg^{-1} (Appendix 1, Table A1.2 and A1.3). The CF of pond aquaculture catfish (Bosma et al. 2011) was larger than net pen aquaculture salmon (Pelletier and Tyedmers 2007). Marine aquaculture and passive gear, such as purse and Danish seine, had the lowest CF for gear and production types (Figure 4.3b).



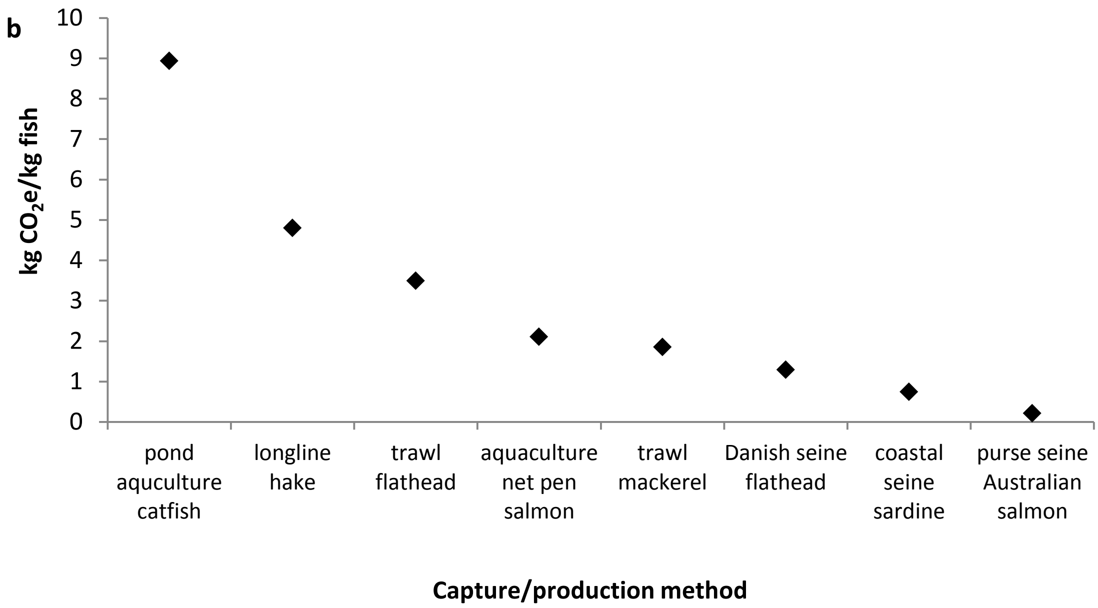


Figure 4.3 Carbon footprint of 1 kg whole prawn and whole fish with different capture and production methods.
See Figure 4.2 for sensitivity of results by species.

4.4.4 Sensitivity analysis

Feed conversion ratio (FCR)

Results for farmed *Litopenaeus vannamei* varied from 2.75 to 6.3 kg CO_2e kg^{-1} (Figure 4.2a) based on the FCR range for semi-intensive and intensive aquaculture presented by Cao et al. (2011). The CF range for *Penaeus monodon* varied from 4.34 to 5.88 kg CO_2e kg^{-1} based on ±15% to account for potential changes in fuel use for collecting snails for feed (Baruthio et al. 2008) (see Appendix 3, Table A3.7). The ranges presented remain comparable to wild-capture *Fenneropenaeus merguiensis* and substantially lower than *Penaeus esculentus*. The standard deviation of FCR for *Pangasianodon hypophthalmus* (Bosma et al. 2011) was used to determine a range of carbon emissions. When ranges are considered for all fish species, the production of *P. hypophthalmus* remains the most carbon intensive per kilogram. Pelletier et al. (2009) provide a FCR range for salmon which was used to calculate the CF range of 1.78 to 2.42 kg CO_2e kg^{-1} for *Salmo salmar*.

Catch per unit effort (CPUE)

CPUE for tiger prawns at the time of the study was 0.15 t/day, which was also the mean CPUE from 2004-2013. Results indicate that when CPUE is higher than 0.15, *Penaeus esculentus* remains more carbon intensive per kilogram than other prawn species. CPUE for *Fenneropenaeus merguiensis*, 1.96 t/day, was higher than the mean for 2004-2013 of 1.5. The CF of 4.2 kg CO_2e kg^{-1} used here is therefore slightly lower than the average for the past decade although the CF for *F. merguiensis* remained similar to that of aquaculture prawns.

The CPUE for *Jasus edwardsii* is at an 11-year low and the CF presented here is higher than it may have been in previous years. The CF of *J. edwardsii* with high CPUE remains larger than *Homarus americanus,* however, the footprints are more comparable when CPUE for *H.americanus* is low (Figure 4.2b).

When the CF range was examined for all fish species, *Merluccius merluccius*, *Neoplatycephalus*, *richardsoni*, *Trachurus trachurus* and *Salmo salmar* were more carbon intensive per kilogram than small pelagics and less than *Pangasius*. CPUE for *Trachurus trachurus* varied from 0.2 to 7.8 t/fishing trip between 1995-2012 resulting in a large range in the CF of 1.85 kg CO_2e kg^{-1} presented here.

System boundaries

The CF varied between 3.8 – 5 CO_2e kg^{-1} for wild-caught banana prawn and 29 – 39 CO_2e kg^{-1} for wild-caught tiger prawn depending on the assumptions made about the inclusion of refrigerants and fuel use for freezing (Appendix 4, Table A3.9). The range of results averaged across the two wild-capture fisheries was 16 - 22 CO_2e kg^{-1}. The inclusion of on-farm activities for net-pen salmon resulted in a 6% increase in carbon emissions and the CF at production rose by 1.3 kg CO_2e kg^{-1} (Appendix 3, Table A3.10), which was higher than the range presented in Figure 4.2c.

4.5 Discussion

4.5.1 Carbon footprint of seafood

The results show that seafood imported into Australia does not necessarily have a higher carbon footprint (CF) than domestically produced seafood, despite the increased distance between production and consumption. It reiterates previous research that food miles, or distance travelled, are not the most accurate measure of impact (Edwards-Jones et al. 2008; Garside et al. 2008; Hogan and Thorpe 2009; Weber and Matthews 2008; Wynen and Vanzetti 2008; Coley et al. 2013) and that production and transportation mode are more important considerations than distance (Avetisyan et al. 2014). Imported products can have comparable or in some cases smaller CF than domestic products. Seafood produced on the other side of the globe, frozen and shipped, may be the most energy efficient (Tlusty and Lagueux 2009), an important consideration for sustainable food policy.

For example, the CF of USA lobster (*Homarus americanus)* on arrival in Sydney was lower than that of the locally produced Tasmanian southern rock lobster (at landing and after airfreight to wholesale) despite travelling approximately 29,000 kilometres from the East coast of the USA by refrigerated container. In contrast, New Zealand is a major supplier of fresh fish to Australia, and although the two countries are neighbours, much of this food is airfreighted and therefore has a higher footprint than some frozen and processed fish transported from further away by sea.

This concept can be explored through the example of carbon emissions associated with plates of seafood consumed in Australia, consisting of 150g of fish, prawn and lobster meat (Figure 4.4). The footprint of a plate of 150g of Australian banana prawns, southern rock lobster and Australian salmon (50g each of edible meat) is 1.5 to 2.5 kg CO_2e. The footprint of a similar plate of imported seafood including 150g of white-leg shrimp, catfish and American lobster is comparable at 1.5 to 2 kg CO_2e. In another example, the CF of a plate made up of Australian wild-capture tiger prawns, southern rock lobster and flathead is between 4 and 6 kg CO_2e, while the CF of a plate made up of imported tiger prawns, American lobster and sardines is only 1 kg CO_2e. The seafood plates compared in these examples are not perfect substitutes, and a comprehensive comparison should account for other sustainability issues, cost, consumer preference, and include other popular species eaten in Australia as well as variations in CF over time. These examples are used here to demonstrate that for consumers and policymakers concerned about carbon footprints of food, imported seafood can be competitive with domestically produced goods.

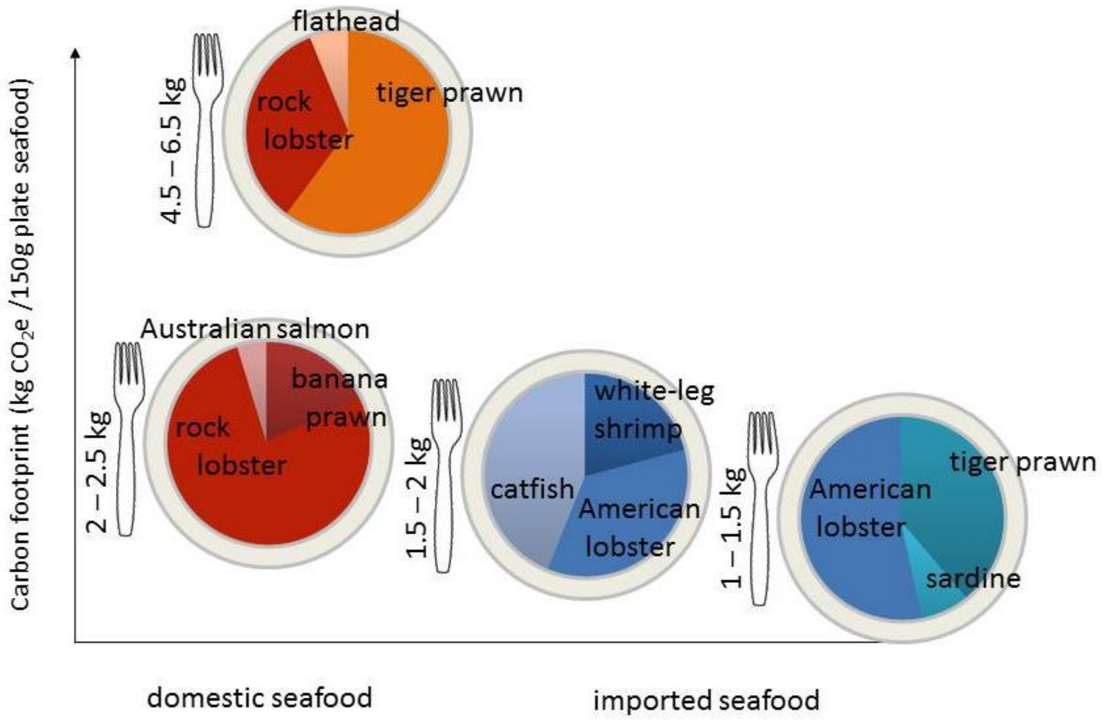


Figure 4.4 Carbon footprints of plates made up of 150g of edible seafood from different domestic and imported sources

Local food production is associated with many positive values (Schnell 2013), and fisheries can potentially be sources of healthy and sustainable local food, in support of the many values and goals embraced by the local food movement (Loring et al. 2013; Kittinger et al. 2015; Carothers and Chambers 2012). However, the use of food miles as a sustainability metric in isolation from additional metrics ignores other supply chain stages and environmental considerations, potentially overshadowing more relevant indicators that are important for balanced debate on food sustainability (Avetisyan et al. 2014). The finding that the production stage (capture or culture), not transport, is typically the major contributor to the CF of seafood products is consistent with the LCA literature (Ziegler and Valentinsson 2008; Thrane 2006; Hospido and Tyedmers 2005; Pelletier and Tyedmers 2010; Vázquez-Rowe et al. 2013; Cao et al. 2011). For wild-capture fisheries, the size of the CF is a reflection of the fuel efficiency of fishing boats, which is determined by the species targeted and the type of gear used (Tyedmers 2001; Schau et al. 2009; Tyedmers and Parker 2012) as well as fisher behaviour and management regime (Vázquez-Rowe and Tyedmers 2013). Management in particular can influence biomass and effort in fisheries which can in turn change fuel use (Chapter 3, Ziegler and Hornborg 2014; Parker et al. 2015; Farmery et al. 2014).

CF of aquaculture species can also vary with farming system. Intensive aquaculture of white-leg shrimp in China, for example, had a higher CF than semi-intensive systems due to higher on-farm energy and feed use (Cao et al. 2011). The degree of intensification, however, may not be as important as other factors such as system efficiency for distinguishing the impacts of aquaculture systems (Aubin et al. 2015). Feed use is a pivotal driver of environmental performance (Pelletier et al. 2009; Henriksson et al. 2014) as seen through large-scale production of tilapia and carp where efficient feeding practices resulted in lower carbon emissions per kilogram than small-scale farming (Mungkung et al. 2013; Henriksson et al. 2015). When production methods are compared, the literature supports the finding here that marine-based aquaculture systems have a comparatively low CF, a function of being less energy-intensive than land-based systems (Ayer and Tyedmers 2009), while pond-based aquaculture can have a higher CF as a result of aeration required to maintain water quality (Pelletier and Tyedmers 2010).

Long-haul airfreighted products have higher CF than non-airfreighted products (Chapter 3, Andersen 2002; Winther et al. 2009; Farmery et al. 2014). Most seafood exported to Australia is sent by sea (www.abs.gov.au) and airfreight is generally required for highly perishable products, where no processing or storage infrastructure exists. Globally, 90% of trade in fish and fishery products consists of processed products (FAO 2012) which negates the need for airfreight and refrigeration, although a trade-off exists where the processing stage contributes to the life cycle impacts of a product. The canning process adopted from Almeida (2015) in this study was the main source of carbon emissions of canned salmon. The footprint of canned products was greater than frozen products predominantly due to the use of tin for the cans. Canning also represented the largest contribution to the CF of tuna (Hospido et al. 2006), anchovy (Avadí et al. 2014) and sardine (Vazquez-Rowe et al. 2014; Almeida et al. 2015) supply chains. The disposal of packaging materials used to ship frozen catfish fillets has also been identified as an area for improvement (Nhu Thuy et al. 2015). Processing, usually nonetheless, prolongs product shelf-life which is an important consideration given that food wastage from storage, handling, transport and final consumption can be as high as 50% for seafood in countries in North America and Oceania (Gustavsson et al. 2011).

Carbon emissions are not a traditional measure of seafood sustainability yet the impacts from climate change, including ocean acidification and rising water temperatures (IPCC 2014), may present a greater threat than the localised production impacts currently informing sustainability assessments. LCA may not capture all of the sustainability issues posed by a globalised, highly complex food system (Garnett 2009) or some unique fishery impacts (Pelletier et al. 2007; Curran et al. 2010), however, opportunity exists to combine the assessment of impacts considered under current measures of sustainability with impacts such as carbon emissions. This combination would provide a more holistic understanding of seafood sustainability as well as highlighting the compatibility or trade-offs between different sustainability goals.

4.5.2 Current seafood sustainability assessment of wild-capture seafood

Demersal trawling can be responsible for ecosystem impacts (Lack 2010) and some of the highest CF of all fishing methods. Opportunity therefore exists to improve the localised ecological impacts of trawling as well as broader environmental impacts through improved fisheries management (Chapter 3, Farmery et al. 2014; Driscoll and Tyedmers 2010; Ziegler and Hornborg 2014). Fuel use efficiency in the Australian Northern Prawn fishery, a global model for many aspects of fisheries management (Gillett 2008), has been improving (Pascoe et al. 2012) although reducing carbon emissions has not been a management goal. Prawn trawl fisheries in Senegal, in contrast, are potentially less well-managed, given that Senegal was ranked alongside the worst performing countries in an assessment of compliance with the FAO Code of Conduct for Responsible Fishing (Pitcher et al. 2008). The CF of trawl caught Senegalese pink shrimp, while not directly comparable to the Australian example, was reportedly high (Ziegler et al. 2011). Management, and its influence on carbon emissions, may be a more important consideration for seafood sustainability than the distance a product has traveled.

4.5.3 Current seafood sustainability assessment of aquaculture

While some farmed seafood can have a lower CF than wild-capture species, there is a range of other environmental impacts associated with aquaculture. Mangrove loss, pollution of agricultural land and water, and impacts on wild fish stocks from wild seed stock collection and feed have all been documented (see for example Páez-Osuna 2001; Diana 2009; Ahmed et al. 2010; Naylor et al. 2000; Jonell and Henriksson 2015). The use of fishmeal has also been identifying as the overall largest single contributor to the CF of the Asian aquaculture sector (Henriksson et al. 2014). However, there have been recent advances in replacement of fishery products in shrimp diets (Glencross et al. 2014).

Several third-party aquaculture assessments have emerged such as the Global Aquaculture Alliance Best Aquaculture Practice certification program. However, the success of the program in reducing environmental impacts is unknown (Tlusty and Tausig 2014). Energy consumption and carbon emissions of farms are included in the Aquaculture Stewardship Council (ASC) standards and a Responsible Feed Standard is currently being developed (asc-aqua.org). This inclusion of the life cycle perspective demonstrates how broader environmental considerations are beginning to be incorporated into seafood sustainability.

4.5.4 Conclusion

Whether seafood products are produced near or far from where they are consumed should not be the main consideration for assessment of their relative sustainability (Tlusty and Lagueux 2009). Instead there needs to be a focus on the whole system - covering production, distribution and consumption. Policy decisions designed to negate the environmental costs of food production through reduced meat consumption, while nourishing a burgeoning populace (Garnett 2009; Eshel et al. 2014), may unintentionally lead to the greater promotion of seafood to meet recommended protein intakes. Policy makers will need to examine existing sustainability criteria, as well as broader impacts associated with species type, production method and distribution mode, while also considering issues of human livelihoods, when considering seafood and sustainability within food policy.

CHAPTER 5: NATURALNESS AS A BASIS FOR INCORPORATING MARINE BIODIVERSITY INTO LIFE CYCLE ASSESSMENT OF SEAFOOD

5.1 Abstract

Food production is a major driver of biodiversity loss and establishing production systems that minimise these impacts must be prioritised. Methods to quantify biodiversity impacts through LCA are evolving for both land- and marine-based production systems, although typically independently from each other. The application of land-based approaches to marine environments is generally not appropriate, and fishery-specific indicators are not widely comparable to terrestrial systems, yet indicators that are applicable across all food production systems are required given their interconnectedness. An indicator for terrestrial food production systems that may be suitable to assess marine biodiversity is a measure of hemeroby, or distance from the natural state. This approach is adapted here to marine systems to assess the impact of fishing on the seafloor and seawater column. The method builds on well-established processes for assessing fisheries within the ecosystem-based fisheries management framework, and is designed to enhance assessment of fishing impacts within LCA and to provide a measure for comparison with other fished and non-fished areas. The method is also a step forward in enabling meaningful comparisons between marine and terrestrial food production systems. A number of challenges were identified through method development and application to case studies and these are discussed, with options for future improvements. Biodiversity is a broad concept not easily captured through a single indicator and this method can complement emerging biotic LCA indicators, to provide a suite of relevant indicators capable of capturing the full impact of fishing on marine biodiversity.

5.2 Introduction

The quantification of seafood carbon footprints, as described in Chapters 2,3 and 4 provides a broader perspective of seafood sustainability, however, results are more meaningful when combined with other indicators including biological ones. Biological impacts are not well integrated into LCA and this chapter explores the development of a new method to assess the impacts of fishing, building on recent developments in fishery-specific LCA methods and on fishery management assessments. Terrestrial and aquatic habitat change have been identified as direct drivers of biodiversity loss, primarily as a result of land use for agriculture and marine use for fishing (Millennium Ecosystem Assessment 2005). Land use is a priority impact category in life cycle assessment (LCA) (Jolliet et al. 2014), and several methods for the quantification of land use in LCA are in advanced stages of development, although a consensus on best practice is yet to be reached (Teixeira et al. 2016; Michelsen and Lindner 2015). Methods to quantify comparable impacts on biodiversity in aquatic habitats have not yet been formalised (Curran et al. 2010). The UNEP/SETAC *Framework for Land Use Impact Assessment within LCA* proposes that physical changes in the seabed be considered as land use related impacts for marine systems, while the biological effects of fishing be considered under the depletion of biotic resources (Milà i Canals et al. 2007) and this approach has been developed further by Langlois et al. (2014a); Langlois et al. (2014b); Langlois et al. (2016). Fishing directly affects both pelagic (water column) and benthic (seafloor) ecosystems (Halpern et al. 2008), and while land is considered an essential support of terrestrial ecosystem services, some life support functions in marine ecosystems are not directly related to the seafloor (Charpy-Roubaud and Sournia 1990). Many marine species have facultative, rather than essential, habitat associations where the seafloor is used for many important life processes, but the absence of these habitats does not result in species extinctions (Foley et al. 2012). Assessments of marine biodiversity therefore need to include aspects of quality and functioning of the whole water column, in addition to biodiversity structures such as sea floor habitat (Derous et al. 2007).

5.2.1 Biodiversity and LCA

Land use and conversion leading to loss in species richness is commonly modelled as an endpoint category in LCA to assess impacts on biodiversity (Souza et al. 2015). Biodiversity impact can be measured by counting species, however, the full impact is not always captured using this approach (Coelho and Michelsen 2014; Langlois et al. 2011; Millennium Ecosystem Assessment 2005) and habitat configuration and use intensity-based considerations are omitted (Teixeira et al. 2016) as well as functional ecosystem roles (de Souza et al. 2013). Alternative methods to assess impacts on biodiversity within LCA have examined relative areas used within ecosystems (Michelsen 2008; Curran et al. 2010) and species-area relationships (Chaudhary et al. 2015). Other authors have focussed on the life support function of land by measuring the biological production capacity (Langlois et al. 2011; Libralato et al. 2008; Langlois et al. 2016; Hélias et al. 2014) or on the quality of land using a measure of naturalness (Brentrup et al. 2002; Fehrenbach et al. 2015).

A number of quantitative and qualitative methods have been proposed to quantify impacts of fishing on different aspects of biodiversity within the LCA framework. Ziegler et al. (2003) estimated the area of seafloor swept per kilogram cod using trawl dimensions, average boat speeds and reported fishing effort in the Baltic Sea. Ellingsen and Aanondsen (2006) also calculated total area swept for cod and compared results with land area required to produce grain for chicken and salmon feeds. Results were presented in square meters and did not include damage assessment, thereby implying that the impact of trawling on an area of seafloor was directly comparable to the impact from converting natural terrestrial habitat to monoculture grain production. A measure of total area swept can be more meaningful when fishing data is overlayed on habitat maps to calculate estimates of area affected by fishing that reflect resilience of the habitat (Nilsson and Ziegler 2007). Several studies have since incorporated this method, however, results continue to be presented as area of seafloor impacted and not weighted according to intensity or resilience (Ziegler and Valentinsson 2008; Vázquez-Rowe et al. 2012b; Ziegler et al. 2011).

A Life Cycle Impact Assessment (LCIA) method to measure the environmental resource footprint of marine area occupation has been proposed for natural marine systems, where the ‘exergy’ content (the maximum work a system can deliver in equilibrium with its environment) of extracted resources is quantified (Taelman et al. 2014). A ‘sea use’ impact category has also been developed to assess transformation and occupation impacts in marine ecosystems as a measure of the free Net Primary Production (fNPP) (2011; Langlois et al. 2014b; Langlois et al. 2016). Quantifying the amount of primary production required to produce seafood, and the pressure placed on ecosystems by overfishing, has been explored by several authors, however, current practice typically does not consider species- and ecosystem-specific factors (Cashion et al. 2016). This approach also faces the challenge that seafood production can rely on primary production from areas distant to the location of the harvest, such as where the production of bivalves relies on primary production of microalgae carried by currents. Aspects of biomass removal by fishing have been explored in LCA through the quantification of the biomass that would not be produced in the future due to current overexploitation (Emanuelsson et al. 2014), and estimates of bycatch and discards (Vázquez-Rowe et al. 2012c; Ziegler and Valentinsson 2008; Ziegler et al. 2011). These approaches to biomass removal also have complexity in their application, for example when fisheries occur with no cost to future production yet still alter the ecosystem from its natural state, for example when a target stock is depleted by fishing but can rapidly recover when fishing stops.

5.2.2 Fishing impacts on seafloor and seawater column biodiversity

There is evidence that the form and function of marine ecosystems can be sustained at a wide variety of fishing pressures, including some cases with very high depletion of the target species, however, the structure of ecosystems under extreme fishing pressure is usually highly modified (Hilborn et al. 2015). Bottom-trawl fisheries impact benthic communities via the dragging of fishing gear over the seafloor and the FAO estimate that 23% of global capture production is obtained from these fisheries (FAO 2009b). Trawling activity has intensified and spread since the 1950s when global records were first assembled (Watson et al. 2006). While bottom-trawling has been likened to clear-felling of forests (Watling and Norse 1998), the effects of trawling vary widely depending on the vulnerability and recovery rates of benthic species and structures (Collie et al. 2000; Kaiser et al. 2006; Althaus et al. 2009). Trawling can cause severe damage to some benthic habitats, such as seamounts (Williams et al. 2010), yet it can be benign on habitats where the benthos is resistant to trawling, particularly in areas where trawl and natural disturbance affect benthic communities in similar ways (van Denderen et al. 2015).

Not all fishing directly impacts the seafloor. Many fisheries operate entirely within the pelagic zone using gear such as purse-seine, pelagic trawl, surface gill-nets and midwater longline, with only incidental impacts on the seafloor through, for example, lost gear and anchoring. This type of fishing has limited impact beyond the capture of species. Although these fisheries barely interact with the seafloor, they nonetheless can affect the naturalness of the system through the removal of biomass. Low fishing pressure typically reduces the average abundance of species without altering ecosystem functions, however, higher exploitation rates can cause changes in trophic structure and very high rates may even lead to depensation and local extinctions (Hilborn et al. 2015). Fishing has also been linked to evolutionary changes in exploited fish stocks, a process which is not currently incorporated into management models (Zimmermann and Jørgensen 2015). Fish stocks do not exist in isolation, and if poorly managed, fishing can place populations of both target and non-target species at risk (ICES 2005). Ecosystem functioning of the whole water column should therefore be considered in assessments of naturalness of the marine environment (Derous et al. 2007), and has been recognised through ecosystem-based fisheries management (EBFM) (Pikitch et al. 2004) and in the assessment approach of independent sustainable seafood certification bodies, such as the Marine Stewardship Council (MSC) (Lack 2004; Mayfield et al. 2014).

5.2.3 'Naturalness' and the hemeroby concept

Natural ecosystems are identified as one of the safeguard subjects, or areas for protection, in LCA (Consoli 1993) and several authors have proposed indicators that incorporate a measure of 'naturalness' of agricultural and forestry production systems within the LCA framework (de Souza et al. 2013; de Baan et al. 2013; Rüdisser et al. 2012; Michelsen 2008). The term *hemeroby* is used in landscape ecology to express distance to nature and has been identified as a consistent method for use in LCA that captures the complexity of land use, with an acceptable level of simplification and without loss of crucial information (Fehrenbach et al. 2015). The hemeroby concept provides a measure of naturalness in a system where the lowest values (ahemerob) correspond to 'natural' or non-disturbed landscapes and the highest values (metahemerob) are given to totally disturbed or 'artificial' landscapes. The hemeroby concept has been adapted for use in LCA to account for the decreasing availability of habitats (Brentrup et al. 2002), to capture the complexity of land use (Fehrenbach et al. 2015) and in combination with other land use indicators (Coelho and Michelsen 2014; Taelman et al. 2016). 'Naturalness' is a complex, multidimensional concept and combining qualitative information on ecosystem quality with quantitative approaches can create a more comprehensive picture (Hochschorner and Finnveden 2003). Several indices have been proposed to express how impacts from agriculture and forestry move the state of the landscape away from the natural one, therefore allowing for characterisation of different types of land use (Machado 2004; Fehrenbach et al. 2015; Brentrup et al. 2002). No such index currently exists to express how impacts from marine production move the state of the ecosystem away from the natural one, although an index for water bodies is reportedly in the early stages of development (Fehrenbach et al. 2015). Adaptation of the hemeroby concept to marine habitats may present an opportunity to overcome the lack of consideration for marine systems in current LCIA methods (Taelman et al. 2014) and provide a more informed comparison of impacts between terrestrial and marine production systems.

5.2.4 Adapting the hemeroby concept to marine habitats

The concept of naturalness can be applied either as a conservation value or as a parameter, or state descriptor, of ecosystems (Machado 2004). The latter is adopted in this research and it is assumed that the closer to ahemerob (natural) a production system the better, in relation to ecosystem functionality and biodiversity, although in terms of food production systems the goal is not to achieve a zero value, as may be the case with a conservation objective. The potential use of the hemeroby concept is examined here as a proxy for the influence exerted by fishing practices on marine biodiversity and the potential application of a Naturalness Degradation Indicator (NDI) using two commercial fishery case studies. A challenge of this research is that marine ecosystems are subject to regime shifts at different time scales (Rocha et al. 2015). This variation through time can complicate identification of the ahemerob (natural) state and original biodiversity. Similar challenges arise in terrestrial ecosystems (Folke et al. 2004) from where the hemeroby method was derived.

We also propose the application of a Naturalness Degradation Indicator (NDI) to semi-quantitatively score the impact of the fisheries on the seafloor and seawater column. Incorporating a measure of marine naturalness into LCA can complement recently developed seafood indicators, such as those addressing impacts from fishing on biomass production capability, using the production target of maximum sustainable yield (MSY), the point at which the highest fish catch can be sustained in the long term, as a reference point (Emanuelsson et al. 2014; Langlois et al. 2014a). Combining indicators that focus on ecosystems, habitats and fish stocks is also consistent with the key framework for land use impact assessment within LCA which recommended that the physical impacts of fishing should be assessed from both 'natural environment' and 'resource' perspectives (Milà i Canals et al. 2007).

The development of this NDI for marine systems is also intended to progress the field of seafood LCA research in a manner that is compatible with more general fisheries management assessments. Much work has been done in the area of ecological risk assessment (ERA) to assess the broader ecological impacts of fishing. For example, the Ecological Risk Assessment for the Effects of Fishing (ERAEF) framework has been applied to over 30 fisheries in Australia and elsewhere and has been adapted for use by the MSC, in particular for use in data-deficient fisheries (Hobday et al. 2011). Current independent seafood certification processes and the ERA process are based around meeting anthropocentric fishery management objectives and the naturalness of the ecosystem and habitat within which the fishery operates is not measured. Here we combine the hemeroby approach with established peer reviewed criteria developed by the MSC with the aim of providing a different, but complementary, perspective which captures the naturalness of marine habitats and ecosystems using a measure of how far removed they are from an unfished state.

5.3 Methods

5.3.1 Index of naturalness

A Naturalness Degradation Index (NDI) is proposed here to classify impacts on marine biodiversity from fishing. The process underpinning application of the NDI, outlined in Figure 5.1, begins with the definition of the area to be assessed. Seafloor and seawater column areas are split at this stage and impacts from fishing defined for each. Available inventory data is defined and used to score areas to a hemeroby class, following a scoring matrix and seven-point scale, to determine how far seafloor or seawater column is from its natural state. Inventory data sources will vary for individual fisheries but may be sourced from resources such as fishery status reports and ecological risk assessments. Impacts on demersal fish that live on the seafloor are captured under the seawater column scale while the impacts to their habitat are captured under the scale for seafloor. For pelagic fisheries that operate in the seawater column only, the assessment can be conducted using the water column scale. For fisheries that interact with both the water column and the seafloor, both scoring scales can be used.

To create the marine hemeroby scale, published scales developed for terrestrial systems (Brentrup et al. 2002; Fehrenbach et al. 2015; Walz and Stein 2014; Steinhardt et al. 1999) were examined and modified based on fishing impacts in marine environments. A seven-point scale is commonly adopted (Walz and Stein 2014) although as the classification of distance to nature can be made at different levels, for example by habitat or land use classes, variations to hemeroby scales can be found in the literature. Depending on the purpose of the study, determination of an areas' hemeroby class may be the final step, or the assessment may progress to the impact assessment stage. This step involves characterisation of the hemeroby score, to determine the Naturalness Degradation Potential (NDP). The NDP is multiplied by the area fished and divided by the functional unit in the final stage of calculating the NDI.

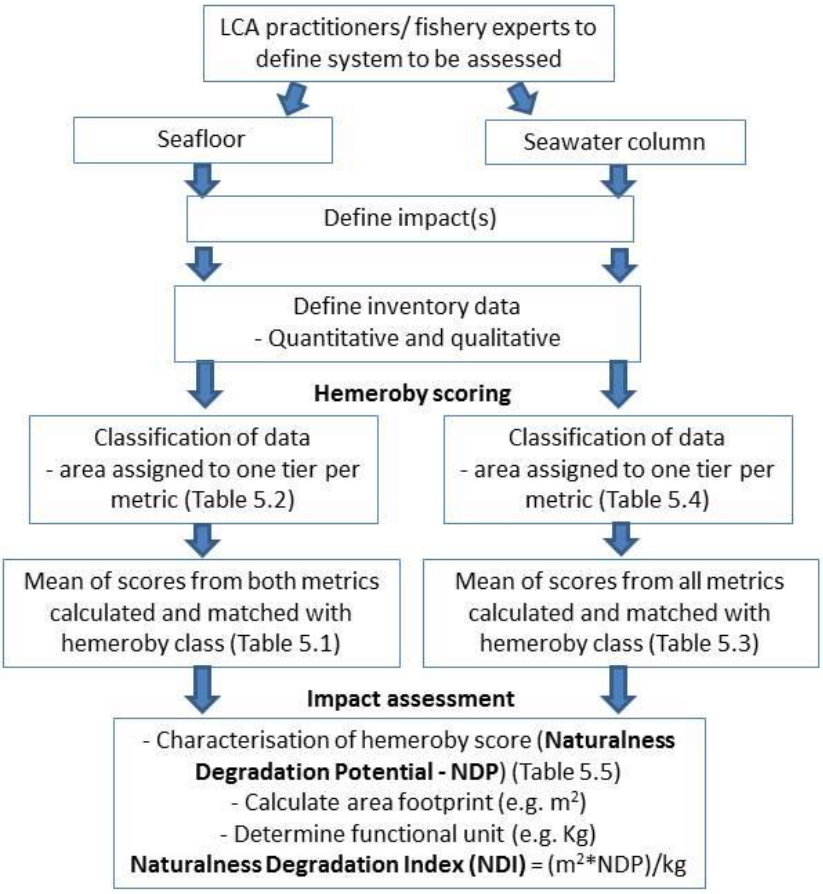


Figure 5.1 Proposed seven step process for allocating seafloor and seawater column areas to hemeroby classes and calculating naturalness degradation impact for marine areas

5.3.2 Fishery case studies

Two Australian fisheries were selected as case studies to demonstrate the application of this method. For a seafloor case study, the Northern Prawn Fishery (NPF), a Commonwealth trawl fishery located in northern Australia, was used. The fishery is currently certified as sustainable by the MSC and not considered to be overfished (Patterson et al. 2015). Otter-trawl gear is used to target a range of tropical prawn species, however, white banana prawn (*Fenneropenaeus merguiensis*) and two species of tiger prawns (*Penaeus esculentus*; *P. semisulcatus*) account for around 80 % of the landed catch (Patterson et al. 2015). The NPF is characterised by a number of benthic habitats including reef platforms, soft sediments and deep siliclastic deposits. The fishery covers an area of approximately 700,000 km^2, however, it is estimated that less than 10% of the total area is trawled (Zhou and Griffiths 2008). Prawn fishing grounds tend to be located on soft sediments that are resilient to perturbation by trawl gear (Bustamante et al. 2010). 2.1% of the total area is never trawled due to permanent area closures, including all shallow water seagrass. Areas that are unsuitable for trawling, such as large reef outcrops and areas with low density of the target prawn species, are also not trawled (AFMA 2013). The impacts of sparse and infrequent trawl effort are not currently considered a threat to biodiversity in the NPF (Pitcher et al. 2009a).

For the seawater column case study the non-bottom trawl Australian sardine (*Sardinops sagax*) fishery was used. Australian sardines are a schooling pelagic fish species found in temperate waters between Rockhampton in Queensland and Shark Bay in Western Australia, including northern Tasmania on the continental shelf. The waters off South Australia support the largest component of the Australian sardine population and sardines have been taken from these waters for use as live bait in the southern bluefin tuna fishery since the 1960s (SASIA 2012). A dedicated purse-seine fishery has been in operation since 1991 and is managed in accordance with a harvest strategy with a set Total Allowable Catch (TAC) and established ecological target and limit reference points (PIRSA 2014). The South Australian sardine stock is currently fished within sustainable limits (Ward et al. 2015).

5.3.3 Seafloor assessment

The scale presented in Table 5.1 was adapted from the literature by the authors in consultation with an expert working group consisting of six scientists from the Institute for Marine and Antarctic Studies (IMAS) at the University of Tasmania and the Australian Commonwealth Scientific and Industrial Research Organisation (CSIRO) with expertise in fisheries ecology and biology, population modelling, and assessment of marine habitats, biodiversity and ecosystem function.. For each hemeroby class on the scale, a description is provided for types of seafloor use with an indicative example. The scale was designed to allow comparison with terrestrial land use.

5.3.4 Criteria and metrics

A classification system has been developed here to score fisheries and assign results to a hemeroby class. Two criteria were defined for implementation of the seafloor assessment. These criteria were based on performance indicators (PIs) developed by the Marine Stewardship Council (MSC 2010) and adapted to reflect naturalness of the seafloor. The MSC has developed a Fisheries Assessment Methodology (FAM) based on three principles: (1) maintaining the productivity of fish stocks; (2) maintaining the structure, productivity, function and diversity of the ecosystem on which the fishery depends; and (3) effective management that meets the requirements of laws and standards and operational frameworks that require responsible and sustainable use of fish stocks (MSC 2002). Each principle is broken down into criteria with specific PIs to be met for certification. For the seafloor scoring metrics the criterion for the 2^{nd} FAM principle and used the 'ecosystems' and 'habitats' criteria were used as these were the most relevant to the seafloor condition (MSC 2010). One metric per criterion was developed each with seven tiers ranging from 0 to 6. These tiers and their descriptions were developed by the authors in consultation with the scientific expert group.

Table 5.1 Definition and description of hemeroby classes for the seafloor

(full scale with land use descriptions in Appendix 4,Table A4.1)

Code	Class	Description and indicative example of seafloor area use and condition
0	Natural	- Seafloor in pristine or near pristine condition subject to only minor indirect influence, e.g. marine debris. Habitats types include highly remote (e.g. oceanic atolls) and very deep habitats (lower continental slope, continental rise and abyss - depths below 2000m) - No fishing influence
1	Close-to-nature	- Seafloor in a natural state and populated by natural species; negligible historical direct impact - Minor (localised or short-term) fishing influence, e.g. anchoring, single line methods
2	Partially close-to-nature	- Seafloor used routinely for human uses - E.g. low to moderate intensity activities e.g. demersal trawl in resilient habitat, Danish seine in unconsolidated sediments
3	Semi-natural	- Intensively used habitats, e.g. historical fishing grounds for bottom contact fishing methods in moderately resilient habitat - E.g. moderate to high intensity demersal trawling
4	Partially distant-to-nature	- Original habitat largely removed, destroyed or permanently altered, especially where there are vulnerable and slowly recovering biota such as large and erect fauna including corals and sponges, and in areas of low productivity including deep continental slopes (depths >200 m); natural biota severely impacted or replaced by invasive or exotic species - E.g. destructive practices: dynamite and cyanide fishing, high-intensity demersal trawl, scallop dredging
5	Distant-to-nature	- No resemblance to original habitat e.g. dredged for sand or highly polluted, with no original biota or communities - Fishing influence prevents regeneration
6	Non-natural artificial	- Reclaimed land with no habitat for marine species, permanent hypoxic 'dead' zones - No relevant fishing influence

As data or evidence of impacts is not always available, a precautionary approach is taken in that fishing activities are assumed to pose higher risks in the absence of information, consistent with MSC (MSC 2010) and ERAEF frameworks (Hobday et al. 2011). Levels of confidence around impacts are incorporated into the assessment to help discern between hemeroby levels. For example, a fishery would be assigned to tier 2 of the ecosystems criterion if there is a high degree of confidence that the fishery is unlikely to cause serious harm, however, if there was low confidence in this impact the fishery would be assigned to tier 3. Conversely, where there is confidence that fishing adversely affects a criterion, the result is allocation to a higher hemeroby class than if there was no confidence around adverse impacts. For example, a fishery would be assigned to tier 4 in the example above if there was evidence that the fishery causes serious harm, and 3 if there was no evidence of serious harm. A clarification of terms used to assess confidence is provided in Appendix 4 (Table A4.3) as well as definitions of other terms used for scoring. The time required for ecosystems to recover is based on their capacity to recover, that is, ecosystems that can recover quickly are deemed resilient and those that are slow to recover are less resilient.

Table 5.2 Seafloor area criteria and metrics – the results for the area under investigation is assigned to the respective tier for each metric

Criterion 1: Habitats	Metric 1: Level and reversibility of harm to habitat structure
	0. No interaction with habitat structure 1. Evidence that there is no harm to habitat structure from fishing 2. Fishery unlikely to cause serious harm, effects are reversible in the short-term, high degree confidence 3. Fishery likely to cause serious harm i.e. alteration of habitat cover/mosaic, effects are reversible in the long-term 4. Evidence fishery causes serious harm, effects are unlikely to be reversible in the long-term i.e. recovery may not automatically occur 5. No resemblance to original habitat 6. Habitat has been completely destroyed
Criterion 2: Ecosystems	Metric 1: Level and reversibility of harm to the key elements of ecosystem structure and function.
	0. Unfished 1. Evidence that fishery does not affect overall biodiversity (species, community composition and structure) 2. Fishery unlikely to cause serious harm, effects are reversible in the short-term, high degree confidence 3. Likely that fishery causes harm, effects are reversible in the long-term 4. Evidence that fishery causes serious harm, effects are unlikely to be reversible in the long-term 5. No resemblance to original ecosystem 6. Ecosystem has been completely destroyed

5.3.5 Seawater column assessment

The scale presented in Table 5.3 was adapted from the literature by the authors in consultation with the scientific expert group to assess the impact of fishing on seawater column area. The scale can be used in combination with the seafloor index for fisheries that impact both the seafloor and seawater column. For each hemeroby class on the scale, a description is provided for the state of the seawater column.

Table 5.3 Definition and description of hemeroby classes for the seawater column (full scale with land use descriptions in Appendix 4, Table A4.2)

Code	Class	Description and indicative example of seawater column area use and condition
0	Natural	- Seawater column in pristine or near pristine condition - no fishing influence
1	Close-to-nature	- Seawater column in natural state, natural species composition - limited removal of species through very low intensity fishing
2	Partially close-to-nature	- Seawater column routinely used for fishing - E.g. low to moderate intensity purse-seine
3	Semi-natural	- Intensively used seawater column - E.g. moderate to high intensity mid-water-trawl
4	Partially distant-to-nature	- Permanently altered seawater column, natural ecosystem severely impacted, especially where there are vulnerable and slowly recovering species, or replaced by invasive or exotic species - E.g. destructive practices or overfishing
5	Distant-to-nature	- Seawater column ecosystem highly modified, no resemblance to original ecosystem e.g. highly polluted, with no natural biota or communities - Fishing influence prevents regeneration
6	Non-natural artificial	- No remaining ecosystem structure or function, e.g. Reclaimed land with no habitat for marine species - No relevant fishing influence

5.3.6 Criteria and metrics

The seawater column scoring metrics were based on criterion from the Marine Stewardship Council's 1st and 2nd FAM principles (MSC 2010). The 'target species', 'retained species', 'bycatch', 'endangered, threatened, protected (ETP) species' and 'ecosystems' PIs were used for metrics as these were most relevant. The 'habitats' PI was not included as this usually refers to bottom habitats, not the water column (MSC 2010). One metric per criterion was developed with tiers applied to each metric to assign the measurement. The authors, in consultation with the scientific expert group, developed seven tiers ranging from 0 to 6 (Table 5.2 and 5.4) and their descriptions.

Table 5.4 Seawater column related criteria and metrics – the results for the area under investigation is assigned to the respective tier for each criterion

Criterion 1: Target species	**Metric 1: target stock biomass** 0. Unfished, 100% virgin biomass (B0) 1. 75-99% B0, high degree confidence 2. 30-74% B0, high degree confidence 3. 20-30% B0 4. Around 20% B0, low degree confidence 5. <20% B0, high degree confidence 6. Species extinct
Criterion 2: Retained/non-target species	**Metric 1: Level and reversibility of harm to the retained species** 0. No retained species 1. Evidence that the level of harm is well below established reference points and has never approached limits 2. Low level of harm, within the limit set by established reference points, high degree of confidence 3. Likely that species could be depleted or recovery hindered– no reference points, low confidence 4. Species seriously depleted and/or recovery hindered, outside established reference points 5. Populations are functionally extinct, no retained species as a result of harm to population 6. Population extinct
Criterion 3: Bycatch and discard species	**Metric 1: Level and reversibility of harm to bycatch species or species groups** 0. no bycatch 1. Evidence that the bycatch level is well below established reference points and has never approached limits 2. Low level of harm, within the limit set by established reference points, high degree of confidence 3. Likely that bycatch species could be depleted or recovery hindered– no reference points, low confidence 4. Bycatch species seriously depleted and/or recovery hindered, outside established reference points 5. Bycatch populations are functionally extinct, no bycatch as a result of harm to populations 6. Populations extinct
Criterion 4: Endangered, threatened, protected (ETP) species	**Metric 1: Level and reversibility of harm to ETP species** 0. No interaction with ETP species 1. Evidence that the risk to ETP species is well below established reference points and has never approached limits 2. Low level of risk, within the limit set by established reference points, high degree of confidence 3. Level of harm likely to impact protection and rebuilding– no reference points, low confidence 4. Level or harm in excess of requirements for protection and rebuilding, risk to ETP species would be high 5. ETP populations are functionally extinct, ETP species not present as a result of harm to populations 6. Populations extinct

Criterion 5: Ecosystems	**Metric 1: Level and reversibility of harm to the key elements of ecosystem structure and function.**
	0. Unfished
	1. Evidence that fishery does not affect overall biodiversity (species, community composition and structure)
	2. Fishery unlikely to cause serious harm, effects are reversible in the short-term, high degree confidence
	3. Likely that fishery causes harm, effects are reversible in the long-term
	4. Evidence that fishery causes serious harm, effects are unlikely to be reversible in the long-term
	5. No resemblance to original ecosystem
	6. Ecosystem has been completely destroyed

Biomass reference points were established for ecosystem thresholds. A common method of specifying biomass reference points is to express them as a percentage of the unfished, virgin biomass (%B0). For target species, 20% B0 is often cited as a default reference point for the minimum acceptable biological limit (Rosenberg 1996; Hilborn and Stokes 2010). However, it is also recognised that the adoption of 20% B0 is unlikely to be applicable across the entire range of observed levels of stock resilience (Hilborn and Stokes 2010). An upper limit of 75% biomass can be used to identify the point at which impacts on trophic structure and ecosystem stability would be small (Smith et al. 2011; Salcido-Guevara et al. 2012). These figures are used as stock thresholds because of their wide application in fishery assessments, however, their use in this context has a different objective, that is, assessing naturalness rather than productivity.

5.3.7 Scoring – seafloor and seawater column

The area being assessed is classified into one tier for each metric (e.g. one tier for habitat, one tier for ecosystem, etc.). The number of the tier represents the same number of points, i.e. classification into tier 1 is associated with one point, tier 2 with two points, and so on following Fehrenbach (2015). The overall score across all metrics is reported as the arithmetic mean. The resulting score between 0 and 6 is then matched with the corresponding hemeroby class (Table 5.1 for seafloor or Table 5.3 for seawater column). Cut-off points were set at 0.5, for example, a score of or between 1.5 and 2.4 will be allocated to hemeroby class 2 (partially close-to-nature). Class 6 may not be reached through fishing but has been included in the index as an indication of activities at that level of impact, following the inclusion of a comparable artificial level in scales developed for agriculture (Fehrenbach et al. 2015; Brentrup et al. 2002).

Hemeroby data can be expressed as ordinal classes, such as those presented in Table 5.1 and 5.3, or as discrete numbers. Life cycle inventory data for naturalness is commonly reported classified according to the respective hemeroby class, although the aggregation into a single indicator value is useful for certain applications (Fehrenbach et al. 2015) and has been proposed for use in LCIA by several authors (Fehrenbach et al. 2015; Brentrup et al. 2002; Taelman et al. 2016). One potential application is an impact assessment of seafood. The process of developing characterisation factors to represent the naturalness degradation potential (NDP) is demonstrated for use in a Naturalness Degradation Indicator (NDI) for marine biodiversity. The NDI calculation is a function of the life cycle inventory data for the area fished (m^2 x 1 year) multiplied by the appropriate characterisation factor (NDP) (Table 5.5) and divided by the functional unit, in this case kilograms of catch ([m^2*NDP]/kg).

5.3.8 Characterisation

In the LCA assessment standard ISO 14044 it is recommended that characterisation factors reflect ‘a distinct identifiable environmental mechanism and/or reproducible empirical observation’ (ISO 2006b). Fehrenbach (2015) allocated factors to reflect the exponentially longer periods of time that natural habitats require to develop. Their approach was also based on current area mix determined through assessments of global land use (UNEP 2014). There is no compatible assessment for the seafloor or seawater column. A linear approach is therefore used, following Brentrup (2002), where intervals between the classes were constant at 0.1665.

Table 5.5 Naturalness degradation potential (NDP) characterisation factors for seafloor and seawater column

Hemeroby class	NDP
0	0
1	0.166
1.8	**0.3 (sardine)**
2	0.333
2.3	**0.37 (prawn)**
3	0.5
4	0.666
5	0.833
6	1.0

5.3.9 Area fished

Three methods were used to calculate bottom trawl ‘footprint’ from trawl effort data for the Northern Prawn Fishery (Pitcher et al. 2016):

- total area of trawl footprint calculated from data gridded at 0.01° within a specified depth-range, covering the total area of cells where trawling was recorded in the past five years.
- a measure of the total swept-area of all trawls annually
- an estimate of annual average footprint based on the total area with trawl and accounting for overlapping effort within grid cells, assuming trawling is conducted randomly at sub-0.01° scale.

Three measures of area fished are compared for the seawater column case study. Area fished was calculated based on the following methods:

- 0.5° cells where fishing had been recorded over the past 10 years (Flood et al. 2014; Emery et al. 2015; SARDI 2016) with fishery specific parameters overlayed using GIS (Table 6). This was the highest resolution available and provides a very coarse measure of area fished.
- annual area fished using data from cells where fishing was reported for 2013.
- annual average number of net sets and gear measurements, where average purse seine net length was 1000m, giving a radius of 159m and an area of approximately 0.08km^2 for each net set. The average number of net sets for the period was 1013 resulting in a value of 81km^2 fished area.

Table 5.6 Fishery specific parameters

Fishery	Parameters used for GIS	Gear and effort
Sardine (*Sardinops sagax*)	- South Australia only - inshore waters to the edge of the continental shelf, down to depths of 200 metres - Exclude reserves and land (Flood et al. 2014; SARDI 2016)	Net length – 1000m Net sets (annual average 2007-2015) – 1013 Net set area – 0.08km^2 (Ward et al. 2015)

5.3.10 Calculating catch

In the case of the NPF, catch was calculated as the average catch over the previous five years (from 2010-2011 to 2014-2015) as annual catch can vary substantially (Woodhams et al. 2011; Woodhams et al. 2012; Woodhams et al. 2013; Georgeson et al. 2014; Patterson et al. 2015).

Australian sardine catch is limited by quotas and the average TAC calculated for the years 2007-2015 (Ward et al. 2015) was used to correlate with area fished data. We also used an example of yearly catch (2013) for comparison with one method of area calculation – the annual average number of net sets and gear measurements (Table 5.7).

5.4 Results

For each fishery case study we calculated three NDI scores based on the different methods for calculating area and catch described in section 5.3.9. The results for the different fisheries are not directly comparable given the different methods used to calculate area.

Table 5.7 Naturalness degradation calculations for seafloor

Fishery	Hemeroby class (calculated using Tables 5.1 & 5.2)	Area with trawling recorded in past 5 years in 0.01° cells (m^2) (Pitcher et al. 2016)	Total swept area (m^2) (Pitcher et al. 2016)	Area of annual average footprint (m^2) (Pitcher et al. 2016)	NDP (Table 5.5)	Catch (kg) 5 year average*	NDI (m^2kg)
Northern Prawn Fishery	2.3	6.79E+07	1.99E+07	1.22E+07	0.37	7.47E+06	3.4 0.98 0.60

*Catch calculated from Woodhams et al. (2011); Woodhams et al. (2012); Woodhams et al. (2013); Georgeson et al. (2014); Patterson et al. (2015).

The seafloor of the NPF was classified as partially-close-to-nature, with a NDP score of 0.37. Trawling in the fishery occurs in resilient habitats and there is a high degree of confidence around the level of impacts with well-defined reference points established for the fishery. The naturalness degradation results varied greatly depending on the trawl area data used. The footprint area based on total area of 0.01° cells with trawling recorded in the past five years was almost six times larger than the footprint area that represented the annual average footprint where overlapping effort within grid cells was accounted for. The NDI results varied from a score of 3.4 m^2 kg^{-1} to 0.6 m^2 kg^{-1} reflecting the different area input data (Table 5.7).

Table 5.8 Naturalness degradation calculations for seawater column

Fishery	Hemeroby class (calculated using Tables 5.3 & 5.4)	Area with fishing recorded in past 10 years in 0.05° cells (m^2)*	Area with fishing recorded in 2013 in 0.05° cells (m^2)*	Area fished based on gear/effort (m^2)*	NDP (Table 5.5)	Catch (kg) (Ward et al. 2015)	NDI (m^2/kg)
South Australian Sardine	1.8	4.14E+10			0.3	3.40E+07 (TAC)	365
			2.82E+10			3.40E+07 (TAC)	249
				8.10E+07		3.40E+07 (TAC)	0.71
				8.10E+07		3.05E+07 (2013)	0.80

*Area calculated using data from Flood et al. (2014); Emery et al. (2015); SARDI (2016)

The seawater column of the South Australian Sardine Fishery was classified as partially-close-to-nature, with a NDP score of 0.3. The biomass of sardines was considered to be between 30-74% based on model-generated estimates of spawning biomass. Schools of sardines are generally highly homogenous and the catch composition of purse seine fishing includes very little bycatch compared to other fishing methods (PIRSA 2014). There is a low level of risk to bycatch and ETP species (SASIA 2012), and retained species (other than sardines). There is also no evidence of ecological impacts from the South Australian Sardine Fishery (Ward et al. 2015). There was some variation in results when using annual or averaged data for both area and catch. The naturalness degradation value of 0.71 $m^2\,kg^{-1}$ for sardine based on the average TAC for 2007-2015 was lower than the value of 0.8 $m^2\,kg^{-1}$ based on 2013 catch, reflecting variability between annual and average catch data when the method of area calculation is the same (Table 5.8). The three examples of area calculation demonstrated that the NDI was very sensitive to the method used, rather than the influence of the fishery. The naturalness degradation value of 365 $m^2\,kg^{-1}$, calculated using the area measurement based on fishing recorded in past 10 years in 0.05° cells, was higher than the value of 249 $m^2\,kg^{-1}$, calculated using the area measurement based on fishing recorded in 2013 in 0.05° cells. The values of 0.71 m^2kg^{-1} and 0.8 $m^2\,kg^{-1}$ calculated using a measure of area based on gear and effort were significantly lower because they did not include the entire region of the GIS grid cell when only a small part of the cell had been exposed to fishing. This measure of area was most similar to the measure of total area swept in the Northern Prawn Fishery.

5.5 Discussion

The hemeroby concept has been developed here as an alternative but complementary approach to including impacts of fishing on biodiversity within LCA. This method for assessing the naturalness of marine systems has been adapted from terrestrial systems to assess the impact of fishing on the seafloor and seawater column. The developed scales are designed to facilitate comparison with land use by terrestrial food production systems, building on published studies (Fehrenbach et al. 2015; Brentrup et al. 2002), and to facilitate greater parity between assessments of marine and terrestrial food production systems. In adapting this method and applying it to our case studies, we identified a range of methodological issues that require consideration and offer some suggestions for consideration for future applications of this method.

5.5.1 Methodological issues

Issues calculating catch and area fished

Calculating annual catches can be complicated in fisheries where catch fluctuates due to environmental or economic reasons. Variance in environmental impacts from one season to another has previously been reported by Ramos et al. (2011) and (Ziegler et al. 2015). For the seawater column, using actual catch or TAC averaged over a number of years, with a measure of area where fishing has been reported over the past decade, was assumed to provide a robust measure of (potential) catch per area. However, without a measure of actual effort this method may seriously over-state the area fished given that a 0.5° cell would be included if it was only fished once with a very small amount of catch. The coarse spatial resolution of fishing effort data has previously been identified as a significant problem for calculating fishing impacts (Nilsson and Ziegler 2007). Applying a lower limit of fishing effort as a criterion for inclusion of cells could be considered in the future.

The case studies reflected the sensitivity of the NDI to choice of data and resolution available as both scored closely in terms of hemeroby class, but had very different impact assessment scores due to the size of the area fished. The influence of area data was also demonstrated within fisheries, for example in the Northern Prawn Fishery the naturalness degradation was lower when the method to calculate average annual trawl footprint was used, rather than the methods using the total of 0.01° cells where trawling had been recorded, or the total area swept, as the annual trawl footprint accounted for overlap. The most substantial difference was in the results for sardine where the method for calculating area had an enormous effect. The naturalness degradation score for sardine when using the area calculation method based on gear and effort was more than two orders of magnitude lower than when using the method for area based on 0.5° cells. The hemeroby score and actual impact of the fishery were overwhelmed by the choices of measurement of area. Naturalness degradation was also sensitive to the use of annual and averaged data and was lower per kilogram of sardine using 2013 data method rather than the 10-year average method, as fishing was concentrated in a smaller area than average for that particular year. Application of the NDI to a wider range of fisheries is recommended to examine the influence of area at different levels of hemeroby. The method used for calculation of area affected is important when comparing naturalness degradation impacts between fisheries, and also with agricultural systems, and care must be taken to ensure that the resolution is comparable. If measurements of area are not compatible across systems, the assessment should not proceed to the impact assessment stage.

Class distribution intervals and characterisation factors

There are several options for the distribution of numerical intervals between each class, including linear constant intervals, exponentially and sigmoid progressing. A linear approach was applied here following Brentrup et al. (2002), however, examining the use of a non-linear approach may be useful for future applications of this method. While the dynamics and stability of natural marine ecosystems is largely unknown, a linear response to environmental drivers has been recorded in marine ecosystems (Lindegren et al. 2016) and populations (Hsieh and Ohman 2006). However, Selkoe et al., (2015) argue that marine ecosystems tend to resist major change until they reach a tipping point. These tipping points can be quantified as zones of rapid change in a nonlinear relationship between ecosystem condition and intensity of a driver. Some marine systems may be prone to tipping points and more information is needed to identify measurable tipping points in the oceans, and the ecosystems that are likely to exhibit tipping points.

Characterising naturalness of systems may also be influenced by social processes. A social response function has been examined in relation to biodiversity offsets where the social process determining the permitted extent of ecosystem service loss over a given time horizon were modelled (Thébaud et al. 2015). Social responses to declining naturalness of systems may follow a sigmoid-shaped curve where moving from a 'natural' state to 'close-to-nature', or from 'distant-to-nature' to 'artificial' is more acceptable than moving between 'partially close-to-nature', 'semi-natural' and 'partially distant-to-nature'. This approach to characterisation has not previously been used in LCA that the authors are aware of, but may potentially be useful in informing marine management and planning and research in the field of social life cycle assessment.

Fehrenbach et al. (2015) developed characterisation factors based on current area mix for terrestrial land use which reflected the effort required to achieve improved naturalness. While a comparable assessment of area mix does not exist for the marine environment, a global map has been developed for human impact on marine ecosystems (Halpern et al. 2008). They found that no area is unaffected by human influence, while a large fraction (41%) is strongly affected by multiple drivers. They also found that large areas of relatively little human influence remained, particularly near the poles. The study accounted for both marine and terrestrial impacts on the marine environment. Further classification of impacts and use in the marine environment could help to inform future development of characterisation factors.

Individual vs collective impacts

Marine areas commonly support several distinct fisheries in terms of the target species and gear used. This situation is in contrast to the terrestrial situation where the land is more likely to be privately owned and managed, and agricultural land uses are likely to be separated spatially, for example cropping and livestock. Assessing one fishery in a multi-fishery zone may result in an NDI score that is not reflective of the current state of the habitat due to greater impacts from another fishery. For example, a trap fishery will have only a small impact on seafloor naturalness, however, if it is operating within a bottom trawl zone the overall state of the area may be far from natural. One way to deal with this situation is to assess all fisheries operating in the area and score the naturalness based on the fishery with the greatest impact. Alternatively, the cumulative score of each fishery can be calculated and would represent the worst possible case. For a single-method fishery targeting multiple species the seafloor impacts would be the same, however, seawater column scoring may vary by species. In this situation, the same approach could be applied as for the seafloor scoring, where the score is based on the species most impacted or on a cumulative score.

The sea use indicator developed by Langlois et al. (Langlois et al. 2014b; Langlois et al. 2016), which accounts for impacts from other human activities in the marine environment, uses free Net Primary Production (fNPP) to expresses the life support capability of the ecosystems. The use of NPP, or primary production required, is emerging as a valuable tool within LCA (Cashion et al. 2016), however, the method does not capture the naturalness of systems and it is possible for managed ecosystems to have a higher ‘productive value’ than natural ecosystems (Taelman et al. 2016). Combining NPP with a measure of naturalness of the system can therefore provide a more holistic assessment of biodiversity impacts from fishing and other human activities. Impacts of cumulative stressors in the oceans has been identified as a top research priority (Rudd 2014) and these types of assessments will become more important as the range and intensity of sea uses increase, including uses such as marine infrastructure, coastal urban development and aquaculture facilities (Dafforn et al. 2015). A current limitation of the method presented here is that it does not capture anthropocentric changes unrelated to fisheries through midpoint indicators, and needs to be combined with methods that do in order to reflect these impacts on marine ecosystems.

5.5.2 Future application

The aim of developing this method is to progress the ability of LCA to provide a measure of equivalent land use that reflects both the area fished and the extent of the damage caused and the ability of the system to recover. Use of this method can enhance assessment of the impact of fishing within LCA to provide a measure for comparison with other fisheries and non-fished areas. Marine environments are largely opaque and changes in marine systems are not as readily visible as in terrestrial systems. This means that causal relationships in marine environments are more uncertain than in terrestrial systems (Johnson and Sandell 2014). The scoring system presented here is novel in its attention to uncertainty. The scales can be used for fisheries where data exists, for example on resilience of habitat assemblages or where limit reference points have been established, as well as for fisheries where data are limited. Most fisheries have adequate qualitative information to enable them to be scored, although lack of data or documentation will result in higher uncertainty about the performance of the fishery (MSC 2015). Greater uncertainty will result in a fishery being classified at a higher hemeroby level (further from natural) than would be the case if more information were available. Providing details on how fisheries are scored is important to ensure transparency of future assessments. For very data poor fisheries with only catch data and type of gear used, using this assessment method may be unfeasible.

Indicators of hemeroby can also be a meaningful supplement to information provided by other national fisheries indicator systems (Walz and Stein 2014). A measure of naturalness can complement established seafood sustainability and marine Ecological Risk Assessments (ERAs) by adding another level of detail, for example, where a fishery is operating at a sustainable level within a permanently altered ecosystem. A measure of naturalness may also provide additional information where current risk of habitat or ecosystem damage is considered low but is a result of prior removal of sensitive species or habitats. The degradation of naturalness of a fishery may also be an important consideration where sustainability assessments are based solely on recent data and the assessment process may be influenced by shifting baselines. In such cases, reliance on recent data can lead to acceptance of the current situation as the natural baseline (Pauly 1995). Using recent data to calculate unfished ecological and stock baselines can be problematic and benchmarking habitat structures (Handley et al. 2014) or drawing on other sources of historical information may be useful to better define the natural state (Pinnegar and Engelhard 2007).

In terrestrial systems, particularly in Europe, the hemeroby concept is well developed and has been used in the field of spatial planning to estimate the cumulative impact of land use changes (Walz and Stein 2014) and to help inform agri-environmental indicators developed for monitoring the integration of environmental concerns into the Common Agricultural Policy (European Union 2012). However, the hemeroby concept has not been applied to marine systems and, as on land, may present a useful method for spatial planning or for informing productivity/environmental indicators, such as those used in EBFM. Incorporating a measure of naturalness of fished areas within planning frameworks can assist with zoning of marine protected areas and in damage assessments by informing trade-offs between development and protection. For example, assessing the naturalness of an area could help inform comparative ecosystem analyses which have been identified as effective methods for use in developing decision support tools for ecosystem-based management of marine areas (Murawski et al. 2010).

Several authors have used the hemeroby approach on the scale of terrestrial bioregions and ecoregions. Data on the type of seafloor substrate and some biome types are strongly lacking at the global scale (Langlois et al. 2016), however, a framework for classifying marine biodiversity on the seafloor has been used for continental-scale bioregionalisation (Last et al. 2010) and may provide a workable basis for defining, managing and conserving biodiversity in the sea at a global scale. Scaling–up the naturalness approach in the marine environment, in combination with these types of assessments, may help inform global analysis on marine ecosystem impacts and help to prioritise management efforts to improve marine ecosystems (Halpern et al. 2015). The sustainability of seafood is also dependant on a range of socio-cultural aspects including the food provisioning functions of small-scale fisheries (Kittinger et al. 2015), resilience of fishing communities, and livelihood options for current and future generations (Lam and Pitcher 2012). Conservation of marine biodiversity, therefore, needs to encompass a range of environmental parameters as well as a range of social-cultural parameters.

5.5.3 Incorporating established frameworks into LCA

There have been calls to incorporate a life cycle approach to management and certification in seafood production for a more holistic sustainability assessment (Pelletier and Tyedmers 2008; Ziegler et al. 2016; Madin and Macreadie 2015; Hornborg et al. 2012), however, there is also merit in using existing indicators and metrics from established seafood sustainability assessment frameworks to inform the development of fishery-specific LCA indicators. A number of independent certification bodies currently assess the sustainability of wild-capture fisheries and species. The Marine Stewardship Council (MSC) is one of the more established seafood ecolabeling programs, with MSC certified fisheries representing approximately 10% of the global harvest of wild-capture fisheries, and over 19,500 products bearing the MSC label in more than 100 countries (www.msc.org). The MSC's Principles and Criteria for Sustainable Fishing were developed through an international consultative process with fishery stakeholders (MSC 2002) and incorporate broader components of ecosystems, including the sustainability of species taken (target and bycatch), as well as the impacts of fishing on other ecologically related species, endangered, threatened or protected species, habitats, and the productivity, diversity, structure and function of ecosystems (Grieve et al. 2011).

Basing the scoring system for hemeroby on established ERAEF and MSC frameworks means that results from all three assessments will have similarities, although they measure different things i.e. risk, sustainability or naturalness. Building on the well-established process for assessing fisheries within the EBFM framework, as has been done here, will help to build more compatible and robust assessments of seafood products within the LCA framework.

5.5.4 Conclusions

Maintaining the ocean's ability to produce food for humans is important given the growing demand for protein combined with increased pressure on land and fresh water resources. Incorporating a measure of naturalness into assessments of food production can be a useful tool to better understand the cost, in terms of transforming ecosystems from natural to more artificial, of meeting the growing demand for food. The hemeroby concept has been used to assess the human impact of food production on land and may offer a useful method for assessing impacts of production in the ocean. However, a number of issues were identified in this study, including the influence of area data used and the need for comparable resolution between studies, which need further consideration for future application to assessments of marine environments.

CHAPTER 6: ASSESSING THE INCLUSION OF SEAFOOD IN THE SUSTAINABLE DIET LITERATURE

6.1 Abstract

The literature on sustainable diets is broad in its scope and application yet is consistently supportive of a move away from animal-based diets toward more plant-based diets. The positioning of seafood within the sustainable diet literature is less clear. A literature review was conducted to examine how the environmental impacts of seafood consumption are assessed and what conclusions are being drawn about the role of seafood in a sustainable diet. Seafood is an essential part of the global food system but is not adequately addressed in most of the sustainable diet literature. Aquaculture, the world's fastest growing food sector, was considered by very few papers. Seafood consumption was commonly presented as a dilema due to the percieved trade-offs between positive health outcomes from eating seafood and concerns of overfishing. A number of studies included seafood as part of their sustainable diet scenario, or as part of a diet that had lower impacts than current consumption. Most of the indicators used were biophysical, with a strong focus on greenhouse gas emissions, and very few studies addressed biological or ecological impacts. The assessment of seafood was limited in many studies due to relevant data sets not being incorporated into the models used. Where they were used, data sources and methodological choices were often not stated thereby limiting the transparency of many studies. Both farmed and wild-capture production methods need to be integrated into research on the impacts of diets and food sustainability to better understand and promote the benefits of sustainable diets.

6.2 Introduction

This chapter builds on the previous chapters by examining how the environmental performance of seafood, determined primarily through LCA, is integrated and interpreted within the rapidly growing body of literature on sustainable diets. The global food system is a major contributor to global environmental change, driven by demand for food from an increasingly larger and wealthier population (Tilman et al. 2001; Godfray et al. 2010). Concern over the environmental impacts of food production, and recognition of the need for more sustainable food systems (International Panel of Experts on Sustainable Food Systems 2015; HLPE 2014a), has driven efforts to measure and compare product environmental footprints to identify opportunities for improvement. Life cycle assessment (LCA) has been widely used to assess the environmental performance of food to better understand a range of environmental impacts generated from the production and supply of food products (Curran 2012). Given that food consumption patterns are a result of both supply- and demand-side factors, a consumption-oriented approach to LCA has emerged, which can complement food product LCAs, to help understand the environmental implications (Heller et al. 2013) and nutritional impacts (Stylianou et al. 2016) of dietary choices. Results from LCAs of specific foods are commonly combined to determine the impacts of whole diets and help promote sustainable patterns of consumption (Hertwich 2005; Girod et al. 2014).

Consideration of environmental impacts in food and nutrition policy is important (Pray 2014; FAO 2010b; Joseph and Clancy 2015) and several European countries (Health Council of the Netherlands 2011; Nordic Council of Ministers 2012; German Council for Sustainable Development 2013), Brazil (Ministry of Health of Brazil 2014), Qatar (Seed 2015) have recently integrated environmental sustainability guidelines into national dietary advice. Efforts to combine advice on health and sustainability in dietary guidelines are still in early development and are not always successful. The Dietary Guidelines Advisory Committee in the USA also recommended that sustainability be taken into account when determining the government's dietary advice (Dietary Guideline Advisory Committee 2015). However, this advice met with opposition (Merrigan et al. 2015) and sustainability was not included in the final 2015 Dietary Guidelines for Americans. In Australia, a section on food, nutrition and environmental sustainability was appended to the Australian Dietary Guidelines 2013 (NHMRC 2013a) following criticism of the inclusion of criteria for environmental sustainability in dietary advice (NHMRC 2013b).

Research on 'sustainable diets' and how modifying consumption patterns can mitigate environmental impacts at both the individual and food system levels, has also increased dramatically in the past decade (Heller et al. 2013; Auestad and Fulgoni 2015; Tilman and Clark 2014; Merrigan et al. 2015; Jones et al. 2016). The FAO defines sustainable diets as those with "low environmental impacts which contribute to food and nutrition security and to healthy life for present and future generations. Sustainable diets are protective and respectful of biodiversity and ecosystems, culturally acceptable, accessible, economically fair and affordable; nutritionally adequate, safe and healthy; while optimising natural and human resources" (FAO 2010c). A sustainable diet consists of several interconnecting components, which have been outlined through a number of conceptual frameworks (FAO 2010c; Johnston et al. 2014; Jones et al. 2016). Food systems are complex social–ecological systems (Prosperi et al. 2016; Tendall et al. 2015) and a simplified summary of these components is presented in Figure 6.1. Due to the broad and interconnected nature of sustainability and human diets, research in this field has evolved along multiple disciplinary lines and is difficult to assimilate due to the disparate frameworks and approaches used (Auestad and Fulgoni 2015). Despite this disparateness, the sustainable diet literature is consistently described as being supportive of a need to move away from animal-based diets toward more plant-based diets (Auestad and Fulgoni 2015; Hallström et al. 2015; Heller et al. 2013; Erb et al. 2016; Meier and Christen 2013). Animal agriculture typically compares unfavourably to plant-based foods due to the additional requirement of converting feed into meat. The feed conversion ratio (FCR), a measure of the quantity of feed required per unit of livestock or aquaculture production, varies substantially between animals. Measures of FCR generally demonstrate that species produced through aquaculture are more efficient converters of feed into animal tissue than poultry, pigs and cows (Forster and Hardy 2001), although some deficiencies have been noted in this measure of efficiency (New and Wijkstrom 1990).

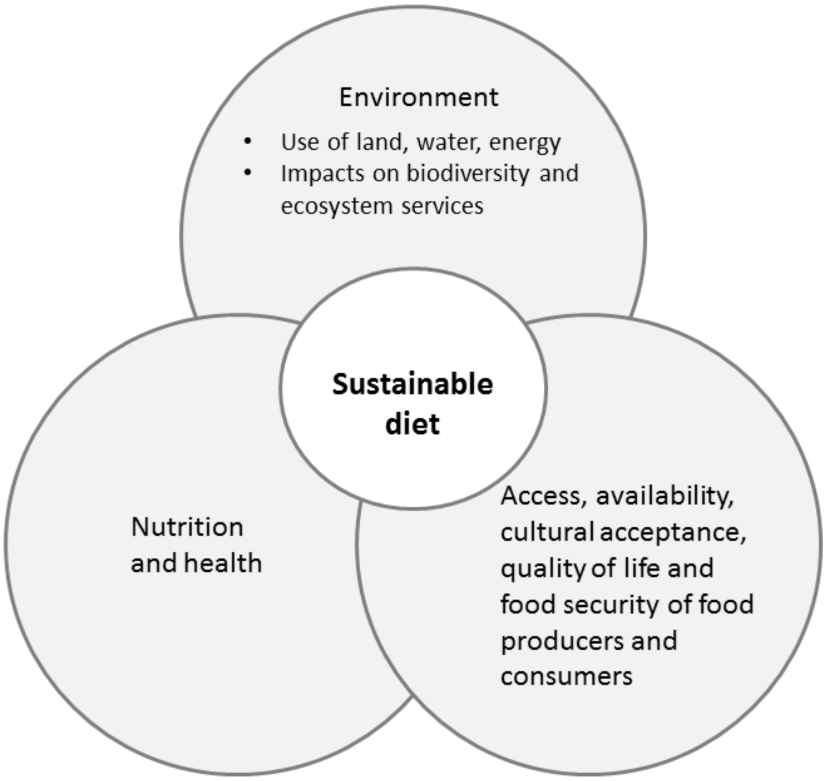


Figure 6.1 Interconnecting components of a sustainable diet showing key elements of environmental sustainability

Seafood (fish and invertebrates from wild-capture fisheries and aquaculture) is an important part of the food system, supplying up to 20% of animal protein intake for more than 2.9 billion people and providing a crucial nutritional component of diets in some densely populated countries where total protein intake levels may be low (FAO 2014b). Seafood is also a source of essential micronutrients, including vitamins D, A and B, minerals (calcium, phosphorus, iodine, zinc, iron and selenium), especially from many small fish species that are consumed whole (HLPE 2014b). Interest in seafood as a source of nutrition historically has focussed on fish oils, as fish are the only major source of the very long chain polyunsaturated fatty acids (VLC-PUFA), eicosapentaenoic acid (EPA) and docosahexaenoic acid (DHA) (Lund 2013), commonly referred to as omega-3 fatty acids. Consumption of marine omega-3 has been linked to protection from heart disease (Lee et al. 2009; Nichols et al. 2010), however, the same health benefits have not been linked to omega 3 supplements (Nestel et al. 2015).

Growth in seafood production currently outpaces population growth (FAO 2014b), with an increasing share sourced from aquaculture, which has been the world's fastest growing food production sector for more than four decades (Tveteras et al. 2012). Global consumption of seafood is approximately 19.2 kg per person, although this amount varies substantially between countries (Smith et al. 2010) and is generally lower than the amount recommended by national dietary guidelines for positive health outcomes (Christenson et al. in press).

While seafood consumption is promoted as part of a healthy diet (Gerber et al. 2012; HLPE 2014b; van Dooren et al. 2014), and it is argued that seafood can continue to make a positive contribution to the food system (Béné et al. 2015; Olson et al. 2014; Frid and Paramor 2012; Troell et al. 2014; Garcia and Rosenberg 2010), the role of seafood in a sustainable diet is less clear. Seafood is regularly excluded from debate on food security (Béné et al. 2015; Thilsted et al. 2016) and food systems research (see for e.g. Erb et al. 2016; O'Riordan and Stoll-Kleemann 2015; Allen and Prosperi 2016; Reisch et al. 2013; McKenzie and Williams 2015; International Panel of Experts on Sustainable Food Systems 2015; Head et al. 2014), despite its substantial contribution to global diets and potential for future growth. This exclusion may reflect the challenge of comparing a traditionally wild-food source with agriculture, and the great variance between marine and terrestrial environments which has resulted in studies on the ecology of these systems developing as largely separate intellectual endeavours (Webb 2012). Concerns over pressure on wild fish stocks have fuelled claims that seafood consumption is not sustainable (Brunner et al. 2009; Selvey and Carey 2013; Jenkins et al. 2009; Greene et al. 2013; Clonan et al. 2012; Thurstan and Roberts 2014) and dietary recommendations for fish intake have been described as the most widely recognised conflict between health and environmental sustainability (Macdiarmid 2013).

In view of the perceived conflict between consuming seafood for positive health outcomes and concerns of overfishing, and given the historical exclusion of fisheries and aquaculture from food system discourse, the published literature on sustainable diets is examined here to determine how seafood is addressed and viewed. Previous reviews of the sustainable diet literature have examined environmental impacts of dietary change (Auestad and Fulgoni 2015; Hallström et al. 2015; Reynolds et al. 2014), and the measurement of sustainable diets (Jones et al. 2016; Heller et al. 2013), although none have specifically reviewed the role of seafood. This paper provides a systematic review of studies that assess the environmental impact of dietary scenarios that include seafood. The objectives are to i) examine how seafood is addressed within this body of literature, in terms of what types of seafood and production methods are included and what impacts are addressed, and ii) summarise the conclusions on the role of seafood in a sustainable diet more broadly. The findings from the sustainable diet literature review are discussed in relation to contemporary research on seafood sustainability. The barriers to, and options for, adequate inclusion of seafood within research on sustainable diets are proposed, as well as opportunities for increased sustainable seafood production.

6.3 Methods

Peer-reviewed published articles included in this review were identified in March 2016 through conventional keyword searching strategies by using Scopus, Web of science and Google Scholar. The search term “sustainable diet$” was used to identify studies published in the past ten years (2010 to 2016). We also identified studies through examination of the references in review articles on sustainable diets (Jones et al. 2016; Hallström et al. 2015; Auestad and Fulgoni 2015; Heller et al. 2013; Reynolds et al. 2014) to capture those studies that may have been missed through use of a single search term. 878 studies were identified in the first stage of the review. The studies were searched for mention of seafood, fish, shellfish, or aquatic products in the context of a sustainable diet in any part of the publication. Of the studies that included these key words (n=504) most (>75%) were excluded as they did not include seafood as an identifiable part of either an actual or modelled diet, for example where seafood was included as part of a ‘meat’ category, or only made reference to seafood or fish briefly in the text. Studies were also excluded if they were not related to human diets, for example those relating to sustainable diets for aquaculture species or if they were not published in English in peer-reviewed journals.

6.4 Results

Forty-seven publications were identified for inclusion in this review (Appendix 5). Publications that met the requirements for inclusion were either quantitative assessments of diets and products (n=32), review articles (n=3) or qualitative discussion papers (n=12). The three review articles identified opportunities and limitations relating to the use of LCA in assessing sustainable diets (Jones et al. 2016; Auestad and Fulgoni 2015; Hallström et al. 2015), therefore these issues are not examined in detail here. However, some fishery and aquaculture specific LCA issues are discussed below.

The focus of this review is on the methods, results and conclusions relating to environmental impacts of diets. The results are presented as follows: brief overview of the methods used in the studies of modelled or actual diets; general results of quantitative studies comparing diets; results relating to individual impacts assessed (GHGe, fresh water, land use, eutrophication, and biological impacts); and general conclusions drawn in the literature from both quantitative and discussion papers.

6.4.1 Methods used in quantitative studies comparing products or diets

Of the 32 studies that included a quantitative assessment of products or diets, 22 were based on process LCAs. In these studies the potential environmental impacts of producing food products was modelled or sourced from published literature. The impacts of individual food items were then aggregated to reflect consumption patterns at the diet or adult meal level. Another of the studies that included modelled or actual diets used the Ecological Footprint (EF) method where a composite indicator is employed to measure the anthropogenic impact by considering the different ways in which environmental resources are used (Ruini et al. 2015). EF is measured in terms of global hectares or square meters, and is calculated as the sum of all the cropland, grazing land, forest, and fishing grounds required to: produce the food and energy required for human activities; absorb all wastes emitted; and provide sufficient space for infrastructure. The authors justified the use of the EF method based on the unit of measure being easier to visualise and understand compared to other indicators, and the methods' ability to consider several environmental impacts simultaneously.

Three further studies used economic input-output analysis as an alternative to process-based LCAs. Hendrie et al. (2014) estimated greenhouse gas emissions (GHGe) for the average Australian adult diet and alternative dietary scenarios using an environmentally extended input-output model of the Australian economy. This method was deemed appropriate by the authors because of the national scale of the study and its focus upon food categories rather than individual food products. Tukker et al., (2011) estimated the difference in impacts between the European *status quo* and three simulated diet baskets using an environmentally extended input-output database, and Weber and Matthews, (2008) used input-output LCA to analyse all relevant emissions of greenhouse gases in the supply chains of food products. The advantages for such an analysis included its ability to handle large bundles of goods, as well as reducing cut-off error, whereby the emissions from processes that are believed to contribute little to the total are excluded, which is considered to be one of the major drawbacks of process-based LCA (Williams et al. 2009). Another challenge of using LCA to compare different products or diets is the influence of methodological choices on results. Differences in choice of functional unit, system boundaries, impact assessment methods and choice of allocation factors can all influence results and should be clarified within the study (Henriksson et al. 2012b). These issues are discussed in more detail in several sustainable diet review papers (Jones et al. 2016; Auestad and Fulgoni 2015; Hallström et al. 2015).

6.4.2 Results of quantitative studies comparing actual and modelled diets

Over half the studies (n=26) assessed seafood as part of an actual or modelled diet (Table A6.1), and a further six made product-based assessments set in a dietary context (Table A6.2) and are discussed under specific impacts below. In a number of the quantitative assessments of actual or modelled diets, seafood formed part of the more sustainable diets, or diets with lower environmental impacts than the average diet (Table 6.1). Diets consisting primarily of seafood and vegetal foods minimised environmental impacts (Gephart et al. 2016) and seafood- and vegetable-rich diets had optimal synergy between health and sustainability (van Dooren et al. 2014). Shifting toward a Mediterranean-type or other more plant-based diets such as pescatarian diets (a diet that includes only vegetables and seafood) had favourable impacts on the environment and health (Table 6.1).

Table 6.1 Summary of diet scenarios examined in quantitative studies and relationships between seafood, environmental performance and health

Diet type	Includes seafood	Reduces impacts from average diet	Meets dietary guidelines	Source
Environmentally sustainable (minimised footprint)	Y	Y	Y	Gephart et al. (2016), Horgan et al. (2016), Hess et al. (2015), Temme et al. (2015), Masset et al. (2014b), Macdiarmid et al. (2012)
	N	Y	N	Donati et al. (2016), Tyszler et al. (2015), van Dooren et al. (2014), Wilson et al. (2013), Vieux et al. (2013), Stehfest et al. (2009)
	N	Y	Y	Fazeni and Steinmüller (2011)
Pescatarian diet/ Mediterranean (high seafood, low meat content)	Y	Y	Y	Scarborough et al. (2014), Ruini et al. (2015), van Dooren and Aiking (2015), van Dooren et al. (2014), Tilman and Clark (2014), Saez-Almendros et al. (2013), Tukker et al. (2011), Eshel and Martin (2006)
Nordic diet (high seafood content)	Y	Y	Y	Röös et al. (2015), Saxe (2014)
Based on dietary guidelines	Y	N	Y	Tom et al. (2016), Tukker et al. (2011)
	Y	Y	Y	Green et al. (2015), van Dooren et al. (2014), Hendrie et al. (2014), Jalava et al. (2014), Stehfest et al. (2009)

Several studies did not include fish as part of their more sustainable diet scenarios, but most of these diets reportedly did not meet national dietary guidelines (Table 6.1). Donati et al. (2016) suggested the complete substitution of meat and fish with vegetal proteins in their dietary model to constitute an affordable and environmentally sustainable diet for young adults, although they noted that from a nutritional point of view this recommendation may not be adequate and a detailed assessment of micronutrients would be required. Similarly, an ‘optimised diet’ which reduced the overall environmental footprint (GHGe, energy and land use) by about 21 % excluded both meat and fish, however, the diet failed to meet the recommendations for intake of omega-3 fatty acids (Tyszler et al. 2015). One study that specifically excluded seafood in their sustainable diet scenario, but managed to meet recommended dietary guidelines, included an increase in consumption of vegetable oil to overcome the lack of omega-3 and 6 fatty acids. It was not clear, however, how much oil would need to be consumed to meet dietary guidelines and if this level of consumption would be realistic (Fazeni and Steinmüller 2011). In their assessment of 16 different diets, Wilson et al. (2013) found that including seafood in a sustainable diet was necessary to meet dietary guidelines for health although the greatest reductions in environmental impacts were made in diets that did not adhere to health guidelines, and may therefore not meet the FAO definition of a sustainable diet as one that is nutritionally adequate.

6.4.3 Contributions to climate change - GHGe

All studies except two examined GHGe, some of them (n=13) as a single indicator (Appendix 6, Table A6.1 and A6.2). Diets rich in fish had lower GHGe than meat diets, but higher than vegetable diets (Eshel and Martin 2006; Saez-Almendros et al. 2013; Tilman and Clark 2014; Scarborough et al. 2014; van Dooren et al. 2014; Vieux et al. 2013). Replacing red meat and dairy with fish, chicken, eggs, or vegetables one day a week was more effective in reducing GHGe than buying locally produced food for one week (Weber and Matthews 2008).

The lower-carbon diets modelled by Masset et al. (2014b) had reduced animal products, including fish. Some seafoods can have moderate to high GHGe in comparison to other food groups (Temme et al. 2015; Green et al. 2015; Tom et al. 2016; Drewnowski et al. 2015), and other seafood, as carbon emissions of different fish and other seafood species vary substantially (Nijdam et al. 2012; Masset et al. 2014a; Carlsson-Kanyama and González 2009; Gephart et al. 2016). Few studies indicated what species were actually included in the seafood category, and whether they were from wild-capture or aquaculture. Examining the original source of the LCA data can help clarify what seafood was examined, however, not all studies indicated the source of the data.

Only three studies examined a range of seafood and provided clear references (Tilman and Clark 2014; Nijdam et al. 2012; Tom et al. 2016). Scarborough et al. (2014) reported emissions for a range of seafood, however, all seafood types were assigned the same value which was sourced from secondary data based on emissions from farmed salmon and trout, imported tuna and shellfish, and UK cod (Audsley et al. 2009).

6.4.4 Energy use

Five studies examined the energy impacts of diets (Table A6.1 and A6.2). Fish consumption was associated with increased energy use as a result of fuel use during fishing (Tyszler et al. 2015) and due to feed production for farmed fish (Tom et al. 2016). In contrast, adoption of the Mediterranean diet, which includes a higher intake of fish than the current Spanish diet, by the Spanish population was estimated to reduce energy consumption by 52% from current dietary patterns (Saez-Almendros et al. 2013)

6.4.5 Fresh water use

Eight studies compared the water footprints of diets (Table A6.1 and A6.2), two of which examined water use as a single indicator (Hess et al. 2015; Jalava et al. 2014). Reducing animal products in the diet offered the potential to save water resources (Jalava et al. 2014; Gephart et al. 2016). Water footprints of fish were low (Tom et al. 2016) or assumed to be zero (Gephart et al. 2016; Hess et al. 2015). Aquaculture was excluded by one study as the required water footprint data were not available (Jalava et al. 2014). A water footprint of seafood was also not available in the database used by Gephart et al. (2016). They instead calculated water use based on global production of the top cultivated aquaculture products (excluding aquatic plants) using the total feeds for each product group, the composition of feeds for each product group, and the water footprint of the inputs. The authors noted that the water footprint of seafood would be higher if all relevant aspects of water use for seafood production were included, such as during processing and evaporative losses from ponds.

6.4.6 Eutrophication

Eutrophication of water and soils was identified as a central issue in animal husbandry and aquaculture (Nijdam et al. 2012), however, this impact category was only addressed by two studies (Tukker et al. 2011; Masset et al. 2014a). Fish was grouped with meat and eggs in the study by Masset et al. (2014a) so it was not possible to determine the contribution of fish to freshwater eutrophication, however, dietary scenarios that reduced eutrophication as a result of reducing the intake of red meat and replacement with chicken, fish and cereals were identified by Tukker et al. (2011).

6.4.7 Land use

Twelve studies that include seafood in their assessments addressed the issue of land use (Table A6.1 and A6.2), however, only three of these studies provided details on land use for the production of seafood (Gephart et al. 2016; Tilman and Clark 2014; Nijdam et al. 2012). Pescatarian diets required less land use than meat-based diets (Tilman and Clark 2014; Gephart et al. 2016). No studies recorded land use for wild-capture seafood although Nijdam et al. (2012) noted that bottom trawling may have an effect on large areas of the seabed. Land use for aquaculture was similar to that of pulses, eggs and poultry (2-6 m^2 y kg^{-1}) (Nijdam et al. 2012). It was unclear if the studies that did not report land use values for seafood assumed no land was used, or excluded seafood from this part of the analysis due to lack of data.

6.4.8 Biological indicators

Only one study addressed biodiversity (Röös et al. 2015) using a measure of biodiversity damage potential (BDP) based on differences in species richness between agricultural and natural land use of the biome. In this study land requirements for food production (m2*year/kg food eaten) were calculated from FAOSTAT and a BDP value from the type of land use (BDP/kg food eaten) was determined. However, no land use or BDP was recorded for fish and no explanation provided. Another study indicated that the model used, E3IOT, was not capable of assessing the impacts on biotic depletion and was thereby not fully able to take into account potential positive or negative impacts of enhanced fish consumption in dietary scenarios (Tukker et al. 2011).

6.4.9 Seafood sustainability conclusions - discussion papers and quantitative assessments

Twelve discussion papers were identified from the literature on sustainable diets that included seafood (Table A6.4), two of which focussed specifically on seafood and sustainable diets (Mitchell 2011; Clonan et al. 2012). Eleven of the studies quantitatively assessing diets or products also provided a discussion on seafood sustainability (Table A6.1 and A6.2). Seven emerging themes were identified (Table 6.2), each of which was mentioned in at least two discussion papers or studies. Although several authors advocated for a greater role of sustainable wild caught seafood, the themes generally reflect quite negative beliefs about seafood and many studies in this literature describe seafood consumption as unsustainable, or present it as a trade-off between health and environmental sustainability (Table 6.2). In their review of the sustainable diet literature Reynolds et al. (2014) concluded that the intake of fish should be reduced in order to reduce the environmental effects of the global diet. Arguments to limit seafood consumption were based on concern that marine fish populations are fully- or over-exploited (Westhoek et al. 2011; Clonan et al. 2012; Lang 2014; Riley and Buttriss 2011) and that aquaculture expansion relies largely on fishmeal, which further depletes fish stocks (Selvey and Carey 2013).

Table 6.2 Themes for seafood identified in the sustainable diets literature

Theme	Source
Dietary recommendations to eat more fish are (potentially) unsustainable	Horgan et al. (2016), Merrigan et al. (2015), Lang (2014), Reynolds et al. (2014), Selvey and Carey (2013), Clonan et al. (2012), Riley and Buttriss (2011), Westhoek et al. (2011)
Consuming seafood presented as a conflict between health and environmental sustainability	Alsaffar (2015), van Dooren et al. (2014), Macdiarmid (2013), Clonan and Holdsworth (2012), Macdiarmid et al. (2012), Mitchell (2011), Riley and Buttriss (2011)
Express concern over environmental/biotic impacts of fishing	Gephart et al. (2016), Tyszler et al. (2015), Ruini et al. (2015), Buttriss and Riley (2013), Heller et al. (2013), Clonan et al. (2012), Nijdam et al. (2012), Tukker et al. (2011), Carlsson-Kanyama and González (2009), Garnett (2011), Mitchell (2011)
Advocate consumption of sustainable wild-capture seafood	Tyszler et al. (2015), Reynolds et al. (2014), Macdiarmid (2013), Buttriss and Riley (2013), Clonan et al. (2012), Riley and Buttriss (2011)
No scope for increased production/consumption	(Fazeni and Steinmüller 2011), (Jalava et al. 2014), (Stehfest et al. 2009)
Use of wild-capture fish for aquafeed should be reduced	Westhoek et al. (2011), Reynolds et al. (2014), Selvey and Carey (2013)
Use of crops for aquafeed will increase footprint of seafood	Westhoek et al. (2011), Gephart et al. (2016)

Several studies with modelled diets did not allow for any future increase in seafood consumption based on the assumption that the oceans are fished to the maximum level, with no capacity for greater wild-fish harvest, and made no allowance for an increase based on growing aquaculture production (Fazeni and Steinmüller 2011; Stehfest et al. 2009; Jalava et al. 2014). Reynolds et al. (2014) stated that growing demand for fish will be met, but only if fish resources are managed sustainably and the animal feeds industry reduces its reliance on wild fish. The reliance of wild fish for aquafeeds was viewed as problematic by several authors (Selvey and Carey 2013; Westhoek et al. 2011). One study suggested that future shifts in the composition of aquaculture feeds away from wild-capture inputs may lead to increased land, water and nitrogen footprints (Gephart et al. 2016). Heller et al. (2013) recommended further examination of the role of sustainable aquaculture in light of the sector's increasing contribution to seafood supply.

6.5 Discussion

One of the biggest challenges for the future food system is the sustainability of protein sources such as meat and fish (Clonan and Holdsworth 2012). The results of dietary comparisons almost unanimously conclude that animal-based foods have greater environmental impact than plant-based foods (Heller et al. 2013). The findings regarding the messages conveyed in the sustainable diet literature relating to seafood consumption support claims that information on seafood sustainability can be conflicting and misleading (Olson et al. 2014). This review of the sustainable diet literature revealed that many studies on the environmental impacts of dietary change are not transparent in their data sources, and include seafood in a manner that reflects neither the large variation within the seafood category nor seafood specific impacts.

6.5.1 Barriers and opportunities to incorporating seafood into sustainable diet research

Not all studies of sustainable diets include seafood (see for e.g. Doran-Browne et al. 2015; Marlow et al. 2015; Goldstein et al. 2016; Sabaté et al. 2015; Kernebeek et al. 2014; Brunelle et al. 2014; Reisch et al. 2013; Temme et al. 2013; Raphaely and Marinova 2014; Erb et al. 2016). The reasons behind the exclusion were not clear, however, a lack of data was cited (e.g. Marlow et al. 2015). A number of studies that did include seafood were also limited by lack of relevant data in standard food databases. The Danish LCA food database was cited by several authors and is one of the only LCA libraries to include data on seafood. Several studies used individual published LCAs to construct averaged data for seafood or relied on external data sets including from the Barilla Centre for Food and Nutrition or from Greenext Service consultants. Consideration of the different impact assessment methods used in LCA, as well as the choice of functional unit, system boundaries, and allocation factors, is essential when comparing LCA results. Evidence of consideration of these important aspects, and uncertainty analysis on how they influence results, was strongly lacking.

Most studies reviewed here did not include details of the contribution of seafood to water footprints. The datasets by Mekonen and Hoekstra (2010; 2012; 2011) were cited by authors comparing the fresh water use of foods, although these datasets do not include seafood. The water footprint of the major farmed species of fish and crustaceans, representing 88% of total fed production has been determined (Pahlow et al. 2015) and future assessments of sustainable diets need to incorporate this type of data.

The source of data for a number of studies was not reported, reinforcing the need for greater transparency around data use in the sustainable diet literature. Reporting the data source is also necessary to ascertain if both wild-capture and aquaculture species are considered in the research. The sustainable diets literature broadly fails to distinguish between seafood on the basis of whether it is wild caught or aquaculture grown, an important consideration given that the main environmental impacts of capture fisheries and aquaculture differ markedly and pose different risks to sustainability of production (Jennings et al. 2016).

Studies identified in the literature review were also limited by models that did not adequately address biological issues, such as the biotic impacts of fisheries (Tukker et al. 2011; Tyszler et al. 2015). The underrepresentation of biological impacts, which are key components of sustainable diets, is not restricted to seafood and has been found across the sustainable diet literature (Jones et al. 2016). Modelling of fishing impacts on stocks and marine ecosystems has advanced in recent years (Plagányi et al. 2014), however, and several marine biotic resource use metrics are under development for use in seafood LCAs (Cashion et al. 2016; Langlois et al. 2014a; Emanuelsson et al. 2014). The sustainable diet literature has failed to keep apace of these developments, presumably as a result of the historical separation of seafood from food system research and discourse, as well as the difficulty in comparing a wild-food source to agriculture, and in applying methods for assessing impacts on land to the sea and vice versa. While biotic impacts of wild-capture seafood can be fishery specific, there is scope for improving comparison across marine and terrestrial systems (Farmery et al. in review; Langlois et al. 2016; FAO 2006). Aquaculture systems may offer more opportunity for comparison with agricultural production, given that the shift toward crop-based feed ingredients fundamentally links seafood production to terrestrial agriculture (Fry et al. 2016), although more research is needed in this area to overcome significant challenges (FAO 2006).

There is a clear need for improved integration of data on the impacts of food production on the land and sea, as well as for methodological standardisation across different production systems. The inclusion of data on a range of wild-capture and aquaculture seafood species in LCA databases should be prioritised and would facilitate the inclusion of seafood in sustainable diet modelling. Data is now available to build a fisheries and aquaculture life cycle inventory library due to the recent growth in seafood LCAs.

6.5.2 Implications of inadequate inclusion of seafood in sustainable diet research

The result of limited access to suitable fishery and aquaculture data is that some researchers modelling future sustainable diets are not allowing for any future increase in seafood consumption (Fazeni and Steinmüller 2011; Stehfest et al. 2009; Jalava et al. 2014) while others refer to seafood only briefly in the context of it being unsustainable (see for e.g. Johnston et al. 2014; Alsaffar 2015; Allen et al. 2014). However, seafood plays, and will continue to play, an important role in the global food system, with annual per capita consumption projected to increase (World Bank 2013). It is imperative that research on sustainable diets incorporates the most efficient and least environmentally damaging products within the seafood category, as within all food categories (Masset et al. 2014a).

Modelling diets on a narrow range of seafood overlooks the fact that wild-capture seafood can have very high or very low GHGe and energy footprints. For wild-capture species the carbon emissions are directly linked to fuel consumption (Avadí and Fréon 2013). Fisheries employing bottom trawls to target crustaceans and flatfish are fuel-intensive, while fisheries targeting small pelagic species such as Peruvian anchovy (*Engraulis ringens*), are the most efficient (Parker and Tyedmers 2014). These low-cost, small pelagic fish are also some of the richest sources of omega-3 fatty acids, however, many are used for non-human uses such as bait or the production of fishmeal and oil due to limited demand for higher-value human consumption markets (FAO 2014b). The opportunity to include these types of seafood in models of sustainable diets is currently being overlooked.

Lack of data on water footprint values for fish (and seafood) was identified as a limitation by Vanham et al. (2013), who substituted meat for fish in their study of potential water saving through dietary change, thereby missing potential water savings from consuming seafood. Wild-capture seafood provides a unique source of food in that it requires little to no freshwater use and no pesticides, fertilisers or antibiotics. The freshwater savings that can be achieved through marine protein consumption (Gephart et al. 2014) are, therefore, also being overlooked in the sustainable diet literature.

The current focus of much of the sustainable diet literature on the unsustainable use of wild fish in aquafeed misses the fact that much of these fish are sourced from well-managed Peruvian anchoveta fisheries which produce some of the least impact-intensive aquafeed ingredients (Pelletier et al. 2009). The replacement of wild-fish ingredients by agricultural products may lead to increased environmental footprints for seafood from aquaculture, as anticipated by several authors (Gephart et al. 2016; Troell et al. 2014; Pahlow et al. 2015) and needs to be included in dietary models. The use of fish processing wastes and land-based by-products for feeds is increasing and will be an important feed ingredient in the future (World Bank 2013). Using waste and by-products, combined with inputs from low-impact, well-managed fisheries, in aquafeeds may present a sustainable option for aquaculture production which does not add to existing impacts from crop and livestock production.

Reducing the amount of fish oil in aquafeed also has implications for the final omega-3 content of the farmed fish, with decreasing EPA and DHA levels recorded in farmed salmon (Sprague et al. 2016; Nichols et al. 2014). While salmon still constitues a good source of fatty acids, larger portion sizes are now required in order to satisfy recommended EPA and DHA intake levels endorsed by dietary guidelines. The shift in fatty acid content was not discussed in the literature examined, however, it is an important element of studies on health and sustainability. Most of the sustainable diet scenarios that did not include fish did not meet national dietary guidelines and may not meet the FAO definition of a sustainable diet as one that is nutritionally adequate. Reduced omega-3 content of aquaculture products may mean that some diets that include seafood may also not meet national dietary guidelines. It should be noted here that some diets can still be associated with positive health outcomes, despite not meeting dietary guidelines, such as vegetarian diets (Ha and de Souza 2015).

This review of the sustainable diet literature revealed that future increases in seafood consumption are frequently viewed as unsustainable, in particular for wild fisheries. However, increasing seafood consumption is not necessarily contrary to good environmental stewardship of the oceans (Mitchell 2011) and debate around the conflict between health and sustainability must also address sustainable pathways for increasing consumption in line with dietary guidelines and growing demand. Highlighted below are some examples of, and opportunities for, increased sustainable seafood consumption.

6.5.3 Opportunities for including wild-capture seafood in future sustainable diets

Eating fish is often presented as a dilemma given that most fished stocks are either fully- or over-exploited (Clonan et al. 2012; Lang 2014; Selvey and Carey 2013; Jalava et al. 2014; Fazeni and Steinmüller 2011; Buttriss and Riley 2013; Riley and Buttriss 2011; Westhoek et al. 2011). It is clear that the opportunities for increasing production in fully-fished stocks are limited, however, the predicted growth in seafood production is anticipated to come from aquaculture and not wild capture fisheries (OECD-FAO 2015). Some opportunities exist to increase the amount of seafood available without increasing catches, such as improved recovery and supply chain management to reduce waste, which can account for up to 50% of edible seafood supply (Love et al. 2015). In addition, the 10% of stocks currently assessed as under-fished and the stocks that are not assessed by the FAO offer potential for increased production. Currently overfished stocks offer another option to increase the amount of seafood available if fishing is properly managed and the stocks are rebuilt (FAO 2014b).

Sourcing seafood from stocks that are widely considered to be sustainable is a priority. Shifting fishing effort away from highly targeted stocks and towards currently underutilised species would reduce pressure on overfished species, result in fewer adverse ecosystem effects of fishing and increase overall fisheries production in the long-term (Zhou et al. 2014b). The transition away from production based on currently overfished stocks may reduce supply in the short term leading to price increases. Demand-side management to support such a transition is needed, such as UK Dietary advice for people who regularly eat fish to consume as wide a variety as possible and experiment with less familiar species from underutilised stocks (Riley and Buttriss 2011). New institutional and market arrangements, such as Community Supported Fishing (CSF) schemes that allow fishers to sell a wider range of species than is currently found in markets (Olson et al. 2014) will also facilitate transition to a lower dependence of seafood on overfished species.

6.5.4 Options for including aquaculture in future sustainable diets

Studies examining current and future dietary scenarios need to address food from aquaculture on an equal basis with crops and livestock and allow for an expansion in seafood consumption, given that aquaculture is the world's fastest growing food production sector. An example of aquaculture being considered in future dietary scenarios is a recent study by Davis et al. (2016), where future growth in seafood demand was met by aquaculture production in their dietary scenarios. Fish from aquaculture have been labelled unsustainable due to the use of wild fish in aquafeeds (Selvey and Carey 2013; Brunner et al. 2009) and there is concern that increasing amounts of fish will be caught for use in aquaculture feeds, to expedite the sector's expansion (Naylor et al. 2000). Yet, despite the growth in aquaculture production, demand for fishmeal and oil has remained steady or declined slightly in recent years (FAO 2011). Demand does not necessarily drive production in wild-capture fisheries, as it does in other food sectors. Increased demand for seafood from fisheries where quotas are set and enforced will generally affect price but not production, particularly where regulation of production in these fisheries is not responsive to market conditions. Seafood from aquaculture production need not be excluded from sustainable diets solely due to the inclusion of wild-fish in feeds.

Not all animals produced by aquaculture are reliant on feed. Bivalves, such as mussels and oysters, use natural ecosystems for food. Production methods requiring little or no feed inputs, such as many bivalve systems, would likely be included more often in minimised diets than seafood as a whole (Gephart et al. 2016). Although bivalve aquaculture presents its own unique impacts, such as the introduction of invasive species (Padilla et al. 2011), they may also have also positive environmental impacts such as reducing eutrophication in waterways and coastal areas (Rose et al. 2014). Polyculture systems also reduce feed use and environmental impact (Neori et al. 2004) while contributing to healthy diets (Thilsted et al. 2016) and may present a sustainable option for future food production.

6.6 Conclusions

The supply of seafood from wild-capture fisheries and aquaculture faces many challenges, as do other food sectors, in order to be considered sustainable. Seafood can provide more sustainable food options than livestock, and in some cases crops, and failing to adequately include seafood in food sustainability, security and nutrition debate risks the promotion of potentially less sustainable and less healthy dietary choices. The debate around seafood consumption needs to shift from a sole focus on biological sustainability to also consider the contribution of seafood to the food system and how to maximise production in the most sustainable manner. Consideration of the sustainability of the linked human systems is also an important component of this field of research. Better inclusion of data on the environmental performance of seafood products in LCA databases, and new methods allowing for comparisons across production systems, are needed to identify diets to meet current and future demand for food with the least environmental impact.

CHAPTER 7: GENERAL DISCUSSION

Seafood sustainability has traditionally been viewed through a resource management lens, and while management of fish stocks and the ecosystems within which they operate is vital, this approach fails to account for the food system impacts generated by seafood. This thesis has examined the sustainability of seafood as an integral part of the global food system as well as a natural resource. Using LCA to assess several wild-capture fisheries and their supply chains enabled quantification of a range of environmental impacts generated during fishing, and beyond the fishery, that are not generally considered in assessments of seafood sustainability. Fishing can result in a broad array of environmental impacts, some of which can be assessed effectively through existing LCA indicators and others which require further development of fishery-specific indicators for application in LCA. Assessing fishing impacts through LCA also highlighted the potential trade-offs that can result from pursuing different sustainability targets. Fuel use during fishing was the major contributor to many life cycle impacts and was greatly influenced by the species targeted, the gear used and the management objectives influencing the fishery. Post-landing activities also contributed substantially to the footprint of seafood products, although these impacts were also influenced by factors such as mode of transportation and should not be viewed in isolation as an indicator of product sustainability. Further development of the LCA method for seafood will help overcome the inadequate inclusion of seafood within research and discussion on sustainable food systems.

Expanding the scope of environmental considerations for seafood by incorporating standardised and emerging life cycle indicators could enhance current assessments of sustainability. It would enable improvements in the fishery, and other supply chain stages, to be monitored over time. Reductions in impacts assessed through LCA may also complement the achievement of other management targets, or capture the trade-offs where they do not. Incorporating a life cycle approach to seafood certification and awareness campaigns can provide consumers with a more holistic assessment of seafood sustainability. For example, in Chapter 2 I showed that while all prawns from the Northern Prawn Fishery (NPF) are considered to be sustainably fished, and are all certified by the Marine Stewardship Council (MSC), fishing for tiger and banana prawns results in very different energy use and carbon footprints and this aspect is not considered by fisheries managers or in the MSC assessment. However, one of the major operators in the NPF, Austral Fisheries, has moved to become carbon neutral since the publication of this research (Figure 7.1).

Figure 7.1 Austral Fisheries' carbon neutral fish logo

Knowledge of the full range of the environmental impacts of seafood production is essential for improving efficiency and developing better practices to limit the footprint of food production (Clay 2011). As the effects of a changing climate begin to affect food production and supply (Wheeler and von Braun 2013), and requirements to disclose carbon emissions and carbon management strategies grow (Jira and Toffel 2013; Matisoff 2013), measuring carbon footprints could become business best practice and an important measure of sustainability.

In Chapter 3, I found that there are likely to be opportunities for revising fisheries management, and more generally marine resource management, to give improved outcomes for both traditional sustainability measures and also a broader suite of environmental impacts. The consideration of environmental metrics provided through LCA can potentially further complicate an already challenging task involving often competing fisheries management objectives and ethical choices such as among employment, economic yield and providing food. Nonetheless, the importance of resource use and emission reduction suggests that fisheries management may need to incorporate a wider range of environmental impacts, in particular carbon emissions and energy use in the future. Fisheries management is adaptive and can change to incorporate a broader range of impacts, as seen through EBFM, although moving from science to policy recommendation to implementation in actual fisheries management can be slow (Skern-Mauritzen et al. 2016).

This research was consistent with the existing seafood LCA literature demonstrating that fuel use by fishing boats is a major contributor to the environmental footprint (Ziegler et al. 2011; Ziegler and Valentinsson 2008; Vázquez-Rowe et al. 2010a; Hospido and Tyedmers 2005). This research also demonstrated that impacts generated outside the fishery can be substantial. In Chapter 3 I showed that the airfreight of rock lobster to China contributed as much as the fuel intensive fishing stage to the carbon footprint, although neither the fuel use in the fishery nor the transport stage are currently considered under sustainability assessments for rock lobster. The transport stage, however, is not always an important contributor to impacts from the export of seafood products. In Chapter 4 I showed that despite increasing distances between production and consumption, the carbon footprints of meals from imported seafood were similar to meals consisting of domestically produced seafood, and sometimes lower, depending on the seafood consumed. Knowledge about seafood production and supply processes is important when assessing the environmental footprint of products (Pelletier and Tyedmers 2008), and basing an assessment on a single stage of the supply chain may not provide a meaningful picture of a products' environmental performance (Weber and Matthews 2008; Coley et al. 2013). Policy makers, therefore, need to re-examine existing sustainability criteria, as well as consider the broader impacts associated with species type, production method and distribution mode, when considering seafood and sustainability within food policy.

Fisheries are an integral part of the global food system, and must be assessed for both their effect on nature and as a source of food. Life cycle practitioners can learn from fishery managers, and established sustainability assessment processes, to further develop biological life cycle impact assessments (LCIA). In Chapter 5 I introduced a new method for assessing impacts of fishing on marine ecosystems. The Naturalness Degradation Indicator (NDI) provides a measure of naturalness of the system as a proxy for impacts on biodiversity. The method highlighted the opportunities to integrate existing fisheries assessments into LCIA, as well as offered scope for comparing results with food production from terrestrial systems. The NDI is also compatible with other recently developed fishery specific indicators (Emanuelsson et al. 2014; Cashion et al. 2016; Langlois et al. 2016). Incorporating a measure of naturalness into assessments of food production is a useful approach to better understand the cost of meeting the growing demand for food, in terms of transforming ecosystems from natural to more artificial. More research on fishery-specific impact categories is required (Avadí and Fréon 2013; Vázquez-Rowe et al. 2012a) for them to be formalised within LCA and to provide a meaningful method for comparing impacts across all food production systems.

Despite the increasing application of LCA to seafood, relevant seafood indicators and data sets have not been adequately incorporated within broader food system assessments. The coverage of seafood within the sustainable diet literature is currently inadequate due to the lack of accessible data and methods, and due to perceptions that overfishing has rendered seafood consumption incompatible with sustainable diets. Seafood can contribute to a sustainable diet, and has the potential to increasingly do so. The debate around consumption, however, needs to move from a sole focus on biological sustainability to also consider the contribution of seafood to the food system and how to maximise production in the most sustainable manner. Opportunities to improve seafood sustainability, and to maximise the potential contribution of the fishery and aquaculture sectors to food and nutrition security, will not be realised as long as seafood is excluded from food systems research and policy (Béné et al. 2015). As long as diets, environmental sustainability and human health are linked, measurement and monitoring of food production and supply will be required to produce the most nutritious food that minimises environmental impacts (Merrigan et al. 2015). A truly sustainable diet will not be achieved unless all aspects of food production from the land and sea are addressed.

This research presents new information on the environmental impacts of seafood as part of the broader food system. I have used Australian examples to examine the environmental performance of selected fisheries and their supply chains and to quantify the trade-offs between environmental impacts from management decisions. This research brings the adequate representation of seafood within food systems research a step closer, an important outcome given the current and potential contribution of seafood to food security and a more sustainable food system. Further development of seafood LCAs and better methods for comparing seafood with agricultural production systems are required to identify the most sustainable production methods and dietary choices. More of this type of research is therefore needed to ensure that seafood, in particular wild-capture production, is included in future food and nutrition security discourse.

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APPENDIX 1

A1 Summary of Life cycle assessment of the Australian Commonwealth Trawl Sector

A1.1 Fishery description

The Commonwealth Trawl Sector (CTS) is one of four sectors in the Southern and Eastern Scalefish and Shark Fishery (SESSF), and is the largest sector in catch and value terms (Perks and Vieira 2010). The fishery is located in waters between Sandy Cape in southern Queensland and Cape Jervis in South Australia (Vieira et al. 2010). More than 100 species of finfish and invertebrates are captured in the sector, although only 20 species are targeted (Smith and Wayte 2005). The five key species that account for the majority of catch are blue grenadier (*Macruronus novaezelandiae*), tiger flathead (*Platycephalus richardsoni),* orange roughy *(Hoplostethus atlanticus*), silver warehou (*Seriolella punctate*) and ling *(Genypterus blacodes)* (Vieira et al. 2010). Of these, blue grenadier, tiger flathead and silver warehou constituted more than 54 % of the 2008–09 catch (Perks and Vieira 2010) and Blue grenadier and flathead were the dominant species in value terms (Woodhams et al. 2011).

The primary harvesting method in the sector is otter-trawling, with a number of Danish seine vessels also operating. A small number of factory trawlers also operate in the fishery, primarily targeting the blue grenadier spawning fishery. The CTS, together with the Scale Hook Sector, are the main source of Australian fresh fish for the Sydney and Melbourne markets.

A1.2 Methods

Data on fuel cost was collected from ABARES (Perks and Vieira 2010) and converted into litres using average price for diesel, minus rebate, for Victoria and NSW. Catch data was sourced through the Australian Fisheries Management Authority (AFMA), subject to a confidentiality agreement. All data relates the period 2005-2006 to 2008-2009 for otter-trawlers and Danish seiners. Freezer boats have been excluded. Data on gear was collected through internet searches and discussion with suppliers and researchers at the Australian Maritime College. Data on resource use at fish landing and processing was sourced from fish co-operatives in Victoria and NSW. Data on wholesale activities was sourced from the Sydney Fish Market.

Appendix 1

The CTS supply chain is represented in Figure A2.1. The system boundary for the LCA included fishing through to secondary wholesale. Inputs for fishing included fuel, gear, and ice, antifoul used on trawlers and Danish seiners. Data collected for wholesale included fuel for vehicles, electricity, packaging, cleaning products and water inputs. Inputs for processing included ice, water, packaging, vehicle fuel and electricity. Data was analysed using SimaPro and the Australian indicator set.

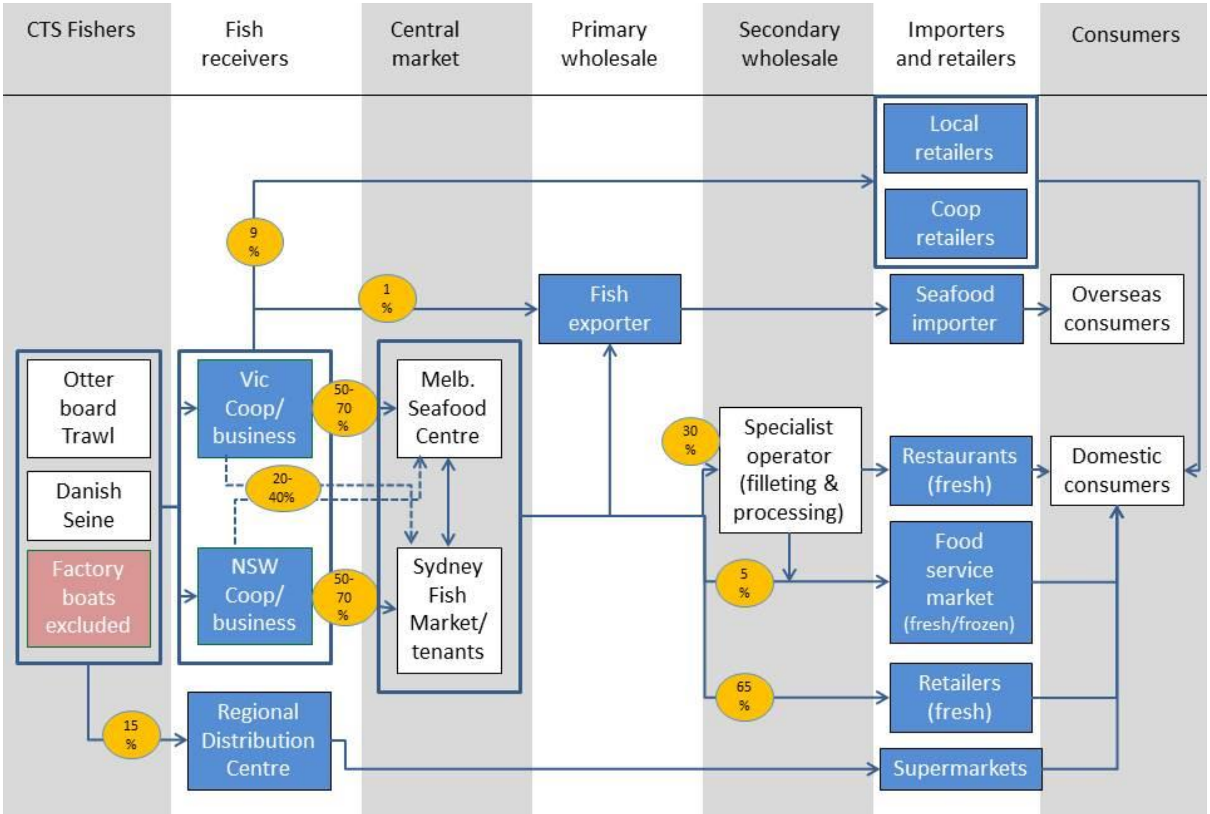

Figure A1.1 Commonwealth Trawl Sector supply chain

A1.3 Results

The fuel used during fishing was the source of the majority of impacts for the global warming potential, cumulative energy use and eutrophication indicators (Table A1.1, Figure A1.2). The processing stage accounted for the bulk of water use per kilogram of fish. This stage also accounted for the greatest share of ecotoxicity, due to emissions from electricity used in this stage. The electricity modelled was a mix of power supplied from New South Wales (NSW) and Victoria. Sensitivity analysis showed that energy from NSW contributed more to the marine aquatic ecotoxicity than energy from Victoria. The GWP for 1 kg of landed fish from the CTS was 2.4 CO_{2e}, based on a weighted average of the number of Danish Seine and otter-trawl boats operating in the fishery.

Appendix 1

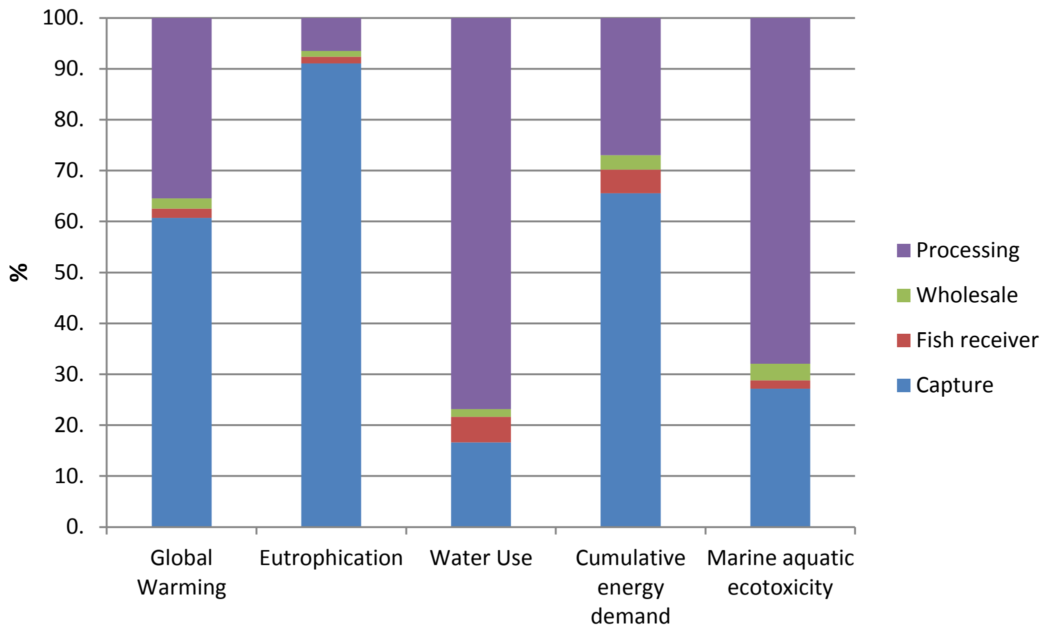

Figure A1.2 Relative proportion of the contribution by impact category to the life cycle impacts of 1 kg of chilled fish from the Commonwealth Trawl Sector

Table A1.1 Life cycle impacts of three stages for 1 kg chilled fish in the Commonwealth Trawl Sector fish supply chain

Impact category	Unit	Capture	Fish receiver	Sydney Fish Market	Processing	Total
Global warming potential	kg CO_2e	2.40E+00	1.00E-01	1.00E-01	1.84E+00	4.44E+00
Eutrophication	kg PO_4e	5.32E-03	9.97E-05	8.64E-05	5.00E-04	6.01E-03
Water Use	M^3	2.57E-03	9.00E-04	3.00E-04	1.36E-02	1.74E-02
Cumulative energy demand	MJ	3.89E+01	3.59E+00	2.16E+00	2.08E+01	6.55E+01
marine aquatic ecotoxicity	DAY	4.52E-11	2.01E-12	4.14E-12	8.56E-11	1.37E-10

Danish seine and otter-board trawl boats have different fuel efficiencies and therefore different global warming potential per kilogram fish caught (Table A1.2 and A1.3). The GWP for Danish seine (1.3 kg $CO_{2e}{}^{-1}$) was lower than for otter-trawl (3.5 kg $CO_{2e}{}^{-1}$).

Table A1.2 Impact assessment 1 kg landed fish from Danish seine

Impact category	Unit	Fuel	Fishing gear	Antifoul	Ice	Total
Global Warming	kg CO_2e	1.19E+00	9.14E-02	1.71E-04	8.47E-03	1.29E+00
Eutrophication	kg PO_4e	2.81E-03	7.29E-05	2.42E-06	1.81E-06	2.89E-03
Water Use	M^3	3.85E-04	3.91E-04	8.56E-06	9.49E-05	8.79E-04
Cumulative energy demand	MJ LHV	1.74E+01	4.17E+00	3.45E-03	9.31E-02	2.17E+01
Marine aquatic ecotoxicity	DAY	8.11E-12	7.64E-12	5.06E-12	7.02E-14	2.09E-11

Appendix 1

Table A1.2 Impact assessment 1 kg landed fish from otter-trawl

Impact category	Unit	Fuel	Fishing gear	Antifoul	Ice	Total
Global Warming	kg CO_2e	3.19E+00	3.13E-01	7.97E-05	8.47E-03	3.51E+00
Eutrophication	kg PO_4e	7.55E-03	2.01E-04	1.13E-06	1.81E-06	7.75E-03
Water Use	M^3	1.03E-03	3.12E-03	4.00E-06	9.49E-05	4.25E-03
Cumulative energy demand	MJ LHV	4.68E+01	9.35E+00	1.61E-03	9.31E-02	5.62E+01
Marine aquatic ecotoxicity	DAY	2.18E-11	4.53E-11	2.36E-12	7.02E-14	6.95E-11

APPENDIX 2

A2 Summary of Life Cycle Assessment of Australian Salmon

A2.1 Fishery description

Eastern Australian salmon (*Arripus trutta*) is targeted in Tasmania by a small number of large vessels specifically equipped to capture and store large quantities of Australian Salmon, and a large number of small vessels which target the species on an opportunistic basis or take them as by-product (Emery et al. 2015). One company accounts for around 85% of landings for the species. This LCA relates to that one company and actual data on inputs is not included for commercial in confidence reasons.

Fish are caught using beach seine gear, which involves deploying a net around a school of Australian Salmon via a small boat and then transferring the catch to the mother ship. Spotter planes are typically used to locate the schools. The majority of the Australian Salmon are frozen whole and sold as rock lobster bait.

A2.2 Methods

Data was collected from the major Australian salmon commercial fisher for 2011. The system boundary included all inputs and activities to the point of landing. Data collection included fuel use by the small and large boats and the spotter plane, as well as for inputs for fishing gear and annual catch. Data was analysed using SimaPro and the Australian indicator set.

A2.3 Results

The largest contributor to all categories was fuel use by the mothership (large boat) (Table A2.1, Figure A2.1), accounting for almost 67% of the global warming potential (GWP) and almost 90% of the eutrophication potential. Aviation gasoline contributed the second greatest amount to GWP and eutrophication. Two-stroke fuel use by the small boat was the second largest contributor to water use and production of lead weights the second largest contributor to ecotoxicity, after the mothership. The fishing net, rope and polystyrene floats made a negligible contribution to any impact category. The GWP of 1 kg of Australian salmon was 0.21 kg CO_{2e}.

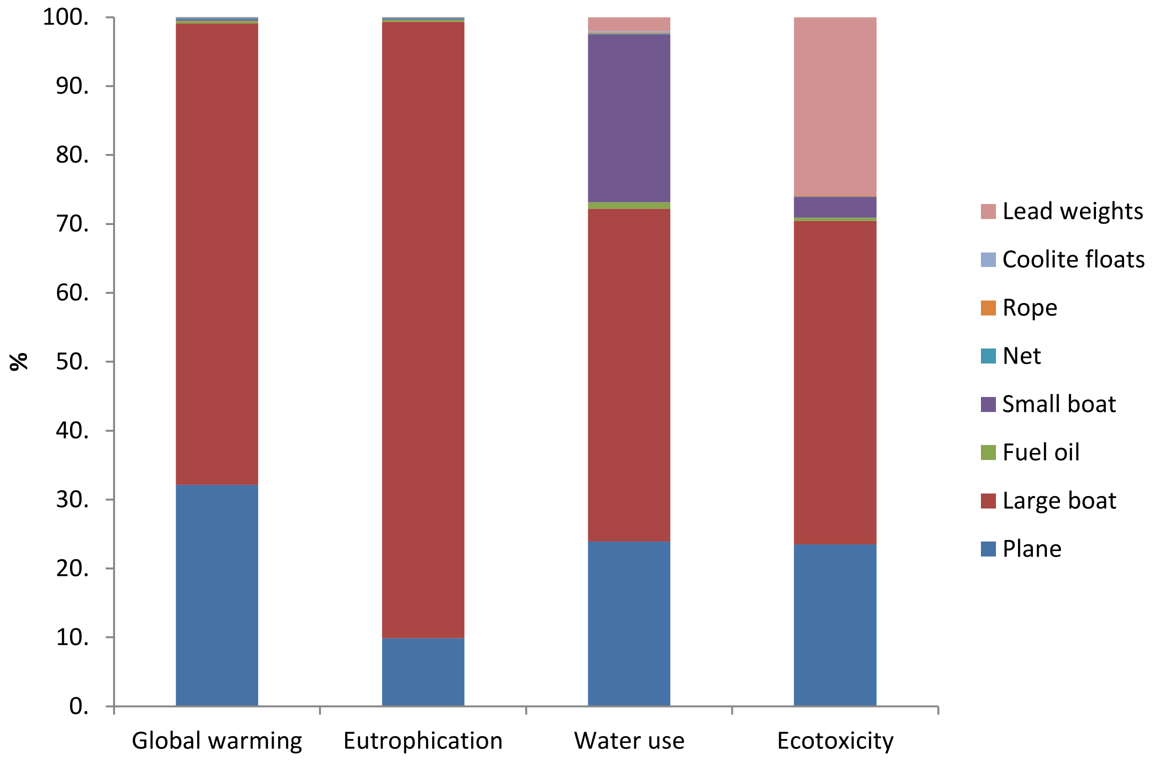

Figure A2.1 Relative proportion of the contribution by impact category to the life cycle impacts of 1 kg of Australian salmon, whole, landed

Table A2.1 Life cycle impacts of three stages for 1 kg Australian salmon, whole, landed

Damage category	Unit	Total	Plane	Large boat	fuel oil	Small boat	Net	rope	Polystyrene floats	Lead weight
Global warming	kg CO_2e	2.07E-01	6.67E-02	1.39E-01	7.75E-04	5.64E-04	7.51E-04	5.13E-05	3.15E-05	3.60E-05
Eutro-phication	PO_4e	4.43E-04	3.76E-05	4.03E-04	9.99E-07	7.09E-07	5.50E-07	1.75E-08	8.02E-09	2.22E-07
Water use		2.29E-02	2.11E-02	1.44E-03	2.43E-04	2.78E-05	1.35E-06	8.90E-08	6.75E-08	2.52E-05
Ecotoxicity	DAY	9.42E+01	7.72E+01	1.58E+01	8.89E-01	1.77E-01	2.28E-02	1.86E-03	9.40E-04	1.27E-01

APPENDIX 3

A3 Methodological issues in LCA comparisons

Comparative LCA studies that include seafood have been used to evaluate the benefits of potential improvements within the same system (Hospido et al. 2006; Papatryphon et al. 2004), to identify the most environmentally-preferred system, method or product (Pelletier and Tyedmers 2007; Ziegler and Valentinsson 2008; Pelletier et al. 2009; Parker and Tyedmers 2012), or choice of management strategy (Chapter 3, Farmery et al. 2014; Driscoll 2008; Hornborg et al. 2012). LCA has also been used to compare products such as anchoveta (Avadí et al. 2014), types of fish and seafood (Aubin et al. 2009; Hall et al. 2011), fish with chicken (Ellingsen and Aanondsen 2006; Mungkung et al. 2012), and seafood with other agricultural products (Foster et al. 2006; González et al. 2011; Mungkung and Gheewala 2007; Nijdam et al. 2012; Tilman and Clark 2014; Hilborn and Tellier 2012; Sonesson et al. 2010). These comparative studies vary from original analysis (see for e.g. Avadí et al. 2014), to recalculation of published results (see for e.g. Nijdam et al. 2012), and reference to published results for comparison (see for e.g. González et al. 2011; Cao et al. 2013).

A3.1 System boundary

The system boundary defines the processes and associated inputs to be included as part of the entire life cycle system. A full LCA covers all life cycle stages from raw material acquisition through to waste disposal, however, in practice most LCAs only include a few of these stages. Data was taken from published LCAs relating to the capture or farm stage only. International air or sea freight to Sydney, Australia, was added based on the most common transport mode for that product. Domestic transport, storage and processing were excluded from the analysis, with the exception of the canning process for salmon which was used as an example of processing impacts in comparison to production and transport stages. Detailed information on the calculation of transport for each case study is presented below. It should be noted that the studies reflect the situation of a particular point in time and results may vary considerably from year to year.

A3.2 Functional unit

The functional unit (FU), or unit of comparison, varied between studies. All FUs were converted to 1 kg of whole product. As the processing stage was not included, except for the example of tinned salmon, 1 kg of seafood at landing or farm gate in this research equates to 1 kg of seafood transported to Australia. Results for canned salmon were calculated back to whole weight. Some seafood was frozen at sea, such as Australian prawns, and the energy use for this stage is included in the CF. Whole products were modelled instead of edible yield as data on processing to produce the edible product was not available for all studies and edible yields are not directly comparable given the seafood types included in the study have different nutritional and economic values.

A3.3 Environmental impacts assessed

Some of the studies used in this research included indicators such as eutrophication potential, energy use and ozone depletion, however, global warming potential was the only indicator consistently used in all papers. The authors recognise that the use of a single indicator does not constitute a full environmental assessment, however, it is useful for creating a better understanding of the link between greenhouse gas emissions and seafood.

A3.4 Allocation

In many cases of food production more than one product is produced (co-production) and the environmental burdens have to be allocated between the different products. Allocation in seafood LCAs, if unavoidable, is typically a choice between economic or mass although other allocation methods have been used. The choice of allocation method can vary results (Iribarren et al. 2010b; Aubin et al. 2015). The studies examined here employ mass allocation, with the exception of Atlantic salmon in which allocation was based on gross nutritional energy content. The authors reported that results from this allocation choice were comparable with mass allocation in this case (Pelletier and Tyedmers 2007).

Table A3.1 Published prawn LCA

Species	Wild/Farmed	Country of origin	Source	Software Database	Characterisation model	Date	Original FU	Allocation	System boundary
Banana prawn (*Fenneropenaeus merguiensis*)	Wild (trawl)	Australia	(Chapter 2, Farmery et al. 2015)	Simapro 7.3.3	CML	2009-2011	1 kg whole frozen	mass	Includes fishing (freezing and packaging)
Tiger prawn (*Penaeus esculentus/semisulcatus*)	Wild (trawl)	Australia	(Chapter 2, Farmery et al. 2015)	Simapro 7.3.3	CML	2009-2011	1 kg whole frozen	mass	Includes fishing (freezing and packaging)
Tiger prawn (*Penaeus monodon*)	Farmed (polyculture)	Philippines	(Baruthio et al. 2008)	Simapro 7.0	CML 2 Baseline 2000	NA	1t fresh aquaculture prawn	mass	Feed for prawns (snails), hatchery, growing out, harvest
White legged shrimp (*Litopenaeus vannamei*)	Farmed (intensive & semi-intensive)	China	(Cao et al. 2011)	Simapro 7.1	CML2 baseline 2000 method (version 2.04).	2008	1t live weight shrimp for cradle-to-farm-gate	mass	Feed production, production of larvae at hatcheries, and production of marketable-size shrimp at the farm level

A3.5 Transport calculation notes for prawns

Table A3.2 sea freight distances of prawns from Results were calculated for the most common route or averaged across several routes from exporting countries. The distance of sea freight to Australia was calculated through ports.com using the average distance from major ports of exporting nation to Sydney, Australia. Journey time was calculated through ports.com/ and chinashippingaustralia.com/pdf/transit.pdf. Ship speed was based on average speed of 10 knots. Freezer operation of a shipping container was based on a standard 40' shipping container, measuring 11.59 m long, 2.29 m width, 2.5 m height with a maximum cargo weight of 26,500kg. The unit for calculation of impacts from the shipping container process was in days. To relate this back to the functional unit of 1 kg, firstly the percentage weight of the container attributable to 1 kg seafood was determined. To do this it was assumed the container was half full to account for container part-loads. 1 kg of prawns accounted for 0.0075% of the total weight. Secondly, 0.0075% of operation time of the container was allocated to the product under examination. Transport within Australia was not calculated for imports or domestic product beyond landing.

Table A3.3 Carbon emissions from production and transport of prawns

Origin	**Production CO_2e kg^{-1}**	**Sea freight CO_2e kg^{-1}**	**Total CO_2e kg^{-1}**	**Transport as % total CO_2e kg^{-1}**
Australia	18			
Philippines	5.1	0.195	5.31	3.68
China (average of intensive and semi-intensive)	5.28	0.23	5.51	4

Table A3.4 Published LCAs of fish consumed in Australia

Species	**Wild/Farmed**	**Landed**	**Source**	**Software/ Database**	**Characterisation model**	**Date**	**Original FU**	**Allocation**	**System boundary**
Flathead (*Platycephalus richardsoni*)	Wild	Australia	Appendix 2	Simapro 7.3.3 Ecoinvent,	CML	2009-2011	1 kg landed	mass	Fishing, ice
Hake *(Merluccius merluccius),* horse mackerel (*Trachurus trachurus*), European pilchard (*Sardina pilchardus)*	Wild	Spain	(Iribarren et al. 2010b)	SimaPro 7	NA	NA	1t landed	Mass	Fishing, refrigerant not included
Salmon *(Salmo salar)*	Farmed	Canada	(Pelletier and Tyedmers 2007)	SimaPro 7.0 various	CML 2 Baseline 2000		1 t of harvest-ready live-weight fish	gross nutritional energy content of the co-products, (comparable to mass)	Feed only
Striped catfish (*Pangasianodon hypophthalmus*)	Farmed	Vietnam	(Bosma et al. 2011)	Simapro EcoInvent®2.0, LCA Food DK	NA	NA	one metric ton (1,000 kg) of fresh fish at the farm's exit-gate (ready for delivery).	mass	Feed, grow out
Australian salmon (*Arripis truta)*	wild	Australia	Appendix 1	Simapro 7.3.3 Australaisian, Ecoinvent,	Australian indicator set – with embodied energy	2009-2011	1 kg whole	mass	Fishing
Sardine *(Sardina pilchardus)*	wild	Portugal	(Almeida et al. 2014)	SimaPro Software version 7.1.6	CML baseline 2 2002	2005 to 2011	1 kg whole sardine landed	mass	Fishing, ice

A3.6 Calculation notes for fish

Data for the canning process of salmon was taken from the canning process for sardines (Almeida et al. 2015). The FU remains 1kg whole salmon which is equivalent to 0.65 kg edible meat, assuming a recovery rate of 65%. This rate is similar for Atlantic salmon and sardines (see www.fao.org/docrep/003/t0219e/t0219e01.htm and www.marineharvest.com/globalassets/investors/handbook/handbook-2014.pdf).

Results from the sardine study were available for 1 kg edible fish and these were recalculated to give results for 1 kg whole fish.

Table A3.5 Published LCAs of lobster consumed in Australia

Species	**Exported from**	**Country of origin**	**Gear**	**Original source**	**Software Database**	**Characterisation model**	**Date**	**Original FU**	**Allocation**	**System boundary**
Jasus edwardsii		Australia	Trap	(Chapter 3, Farmery et al. 2014)	Simapro 7 Ecoinvent	CML	2010/11	1 kg live weight	mass	Fishing, bait
Homarus americanus	Louisville, US	Canada	Trap	(Boyd 2008)	SimaPro 7.1.2 Various	CML	2006/6	1 t live weight	mass	Fishing, bait
Homarus americanus	Nova Scotia, Canada	Canada	Trap	(Boyd 2008)	SimaPro 7.1.2 Various	CML	2006/6	1 t live weight	mass	Fishing, bait
Homarus americanus	Boston, US	United States	Trap	(Driscoll 2008)	SimaPro 7.1.4 EcoInvent, IDEMAT, Franklin	CML Baseline 2000	2006	1 t live weight	mass	Fishing, bait, vessel

For *Homarus americanus* from the USA (Driscoll 2008) airfreight to Sydney from Boston was added, or sea freight from Boston to Sydney.

Table A3.6 Carbon emissions of capture of lobster and transport to Sydney

Species	Capture $CO_2e\ kg^{-1}$	Sea freight $CO_2e\ kg^{-1}$	Air freight $CO_2e\ kg^{-1}$	Total
Australia (Tas) Trap *Jasus edwardsii*	12.4		1.21	13.6
USA Trap *Homarus americanus*	3.27		18.3	21.6
USA Trap *Homarus americanus*	3.27	0.7		3.97

A3.7 Sensitivity analysis

Table A3.7 Feed conversion ratios and reported ranges

Species	Study	FCR range	FCR reported	Comment
Salmon (*Salmo salar*)	(Pelletier and Tyedmers 2007)	1.1 – 1.5	1.3	Range taken from Pelletier et al. (2009)
Chinese shrimp (*Litopenaeus vannamei*)	(Cao et al. 2011)	0.97 – 2.1	1.6	0.97 - semi-intensive, 2.1 intensive system
Catfish (*Pangasianodon hypophthalmus*)	(Bosma et al. 2011)	±0.28	1.86	FCR range based on standard deviation
Tiger prawn (*Penaeus monodon*)	(Baruthio et al. 2008)	±15%	N/A	Main tiger prawn feed consists of horn snails, no FCR provided. Range based on 15% variation of CF assuming increase or decrease fuel use for collecting snails.

Table A3.8 Catch per unit effort (CPUE) at time of study and standard deviation of carbon footprint modelled on CPUE

Species	Study	CPUE	SD carbon footprint	Comment	CPUE reference
Tiger prawn (*Penaeus esculentus*)	(Chapter 2, Farmery et al. 2015)	0.15 t/day (mean 0.15)	±1.07	Mean CPUE 2004-2013	afma.gov.au
Banana prawn (*Fenneropenaeus merguiensis*)	(Chapter 2, Farmery et al. 2015)	1.96 t/day (mean 1.5)	±4.74	Mean CPUE 2004-2013	afma.gov.au
Southern rock lobster (*Jasus edwardsii*)	(Chapter 3, Farmery et al. 2014)	0.79 kg/trap lift (mean 1)	±2.11	Mean CPUE 1995-2012	(Hartmann et al. 2012)
American lobster (*Homarus americanus*)	(Driscoll 2008; Boyd 2008)	1.1 kg/trap lift, Canada (mean 1.35) 13.3 #/m^2, USA (mean 15)	±1.09 (Canada) ±1.1 (USA)	Mean CPUE 2004-2013, range calculated from maximum CF USA and minimum Canada	http://www.dfo-mpo.gc.ca (ASMFC 2014)
Hake (*Merluccius merluccius*)	(Iribarren et al. 2010b)	0.83 t/fishing days (mean 0.93)	±1.72	Mean CPUE 1995-2012	(Castro et al. 2014)
Flathead (*Neoplatycephalus richardsoni*)	Appendix 2	30.8 kg/hr trawl (mean 30.2) 96.2 kg/shot Danish seine (mean 90.3)	±0.17 (trawl)±0.18 (Danish seine)	Mean CPUE 2005-2013	(Sporcic and Haddon draft)
Horse mackerel (*Trachurus trachurus*)	(Iribarren et al. 2010b)	1.3 t/fishing trip (mean 1.9)	±2.86	Mean CPUE 1995-2012. 2005 excluded	(ICES 2013)
Sardine (*Sardina pilchardus*)	(Iribarren et al. 2010b; Almeida et al. 2014)	0.69 t/fishing trip (mean 0.79)	±0.76	Mean CPUE 1995-2012	(ICES 2013)
Australian salmon (*Arripus trutta*)	Appendix 1	16 t/fishing day (mean 19.2)	±0.06	Mean CPUE 1995-2012	(André et al. 2014)

Prawn

The functional unit for prawn from the Northern prawn fishery was 1 kg frozen whole prawn at landing as all prawns are frozen at sea. The CF of these prawns at landing therefore includes on-board freezing, an input that is not included in the aquaculture prawn studies. The principle driver of energy use in fishing operations is the burning of fuel for propulsion and gear operation (Thomas et al. 2010), however, fuel use for refrigeration and freezing can account for between 3 and 12.5% of total fuel use (Wakeford 2006). Sensitivity analysis was therefore conducted for the energy used for on-board freezing. Assuming 10% of fuel use went to freezing, results were decreased by 10% (Appendix 3, Table A3.9).

Use of refrigerants was not included in the assessments however refrigerant leakage increases the GWP of wild capture seafood between 13 – 20% (Vázquez-Rowe et al. 2012b, 2010b; Iribarren et al. 2011). Sensitivity analysis was therefore carried out. Results for wild capture prawns were increased by 20% to account for the maximum possible impact of refrigerants. The results show that the scenarios do not alter the ranking of species in terms of carbon emissions per kilogram, although actual kilograms of carbon emitted do vary. The CF for wild capture tiger prawns remains much higher than for wild capture banana prawns or aquaculture species. For banana prawns, the CF remains similar to aquaculture species.

Table A3.9. Carbon emissions for wild capture and aquaculture prawns with the addition of emissions for refrigerants (20%) and reduction of emissions from freezing (10%) for wild capture prawns.

Origin	**Method**	**Species**	**landed frozen/live weight (kg CO_2e)**	**-10% freezing excluded (kg CO_2e)**	**+20% refrigerants included (kg CO_2e)**
Australia	Trawl	*Penaeus esculentus*	32	29	39
China	Aquaculture (intensive)	*Litopenaeus vannamei*	5.3		
Philippines	Aquaculture (polyculture)	*Penaeus monodon*	5.1		
Australia	Trawl	*Fenneropenaeus merguiensis*	4.2	3.8	5
China	Aquaculture (average intensive and semi-intensive)	*Litopenaeus vannamei*	3.1		
China	Aquaculture (semi-intensive)	*Litopenaeus vannamei*	2.8		

Results for the carbon footprint of banana prawn varied between 3.8 – 5 CO_2e kg^{-1}, depending on the assumptions made about the inclusion of refrigerants and fuel use for freezing. Sensitivity analysis for tiger prawns resulted in a range between 29 – 30 CO_2e kg^{-1} which is still much higher than other prawns examined. The range of results averaged across the two wild capture fisheries was 16 - 22 CO_2e kg^{-1} with sensitivity analysis.

Fish

The CF for flathead and Portuguese sardine includes impacts from ice used to cool fish, while neither ice nor refrigeration are included for hake, mackerel or European pilchard. For flathead and Portuguese sardine, ice contributed less than 1% to CF at capture (Appendix 1, Almeida et al. 2014) therefore sensitivity analysis was not conducted. Neither ice nor refrigeration is included in the study for hake, mackerel or European pilchard.

Capital goods were excluded from the CF of wild capture fish with the exception of trawl gear for flathead. Trawl gear accounted for less than 10% of the CF of both otter-trawl and Danish seine caught flathead (appendix 1). Burdens related to gear have also been found to be low for octopus (Vázquez-Rowe et al. 2012a). Capital goods were excluded from assessments in accordance with the guidelines provided by PAS 2050 (BSI, 2008) and following previous findings that they make a minor contribution to the overall environmental impacts of fisheries and seafood products (Nijdam et al. 2012; Ziegler et al. 2013).

The CF of farmed catfish included production of feed and grow-out while the CF for salmon included feed only. Feed production is the major source of cradle-to-farm-gate life cycle impacts for farmed salmon production, including 94% of global warming emissions (Pelletier et al. 2009). Sensitivity analysis was performed to test the influence of including the on-farm activities for net-pen salmon. Results were increased by 6% to account for these activities and the CF at production rose by 1.3 CO_2e kg^{-1} (Table A3.10).

Lobster

Vessel production was included in the study of lobster from the US (Driscoll and Tyedmers 2010) but excluded from other lobster studies. As stated above, emissions from capital goods such as vessels make a minor contribution to the overall environmental impacts of fisheries and seafood products. Refrigeration was not considered during fishing as no boats used on-board cooling.

Table A3.10. Carbon emissions for different salmon products at production, processing and transport, with the addition of 6% $CO_2e\ kg^{-1}$ at production to account for grow-out stage

Fish (*Salmo salar)*	**Origin**	**Production** $CO_2e\ kg^{-1}$	**Sea freight** $CO_2e\ kg^{-1}$	**Air freight** $CO_2e\ kg^{-1}$	**Canning** $CO_2e\ kg^{-1}$	**Total** $CO_2e\ kg^{-1}$	**Production % total**	**Transport % total**	**Canning % total**
Frozen salmon	USA	2.1	0.7			**2.8**	75	25	
Canned salmon	USA	2.1	0.3		6.6	**9**	24	3	73
Fresh salmon	USA	2.1		18.3		**20**	10	90	
Frozen salmon +6%	USA	3.4	0.7			**4.1**	79	21	
Canned salmon +6%	USA	3.4	0.3		6.6	**10**	33	3	64
Fresh salmon +6%	USA	3.4		18.3		**22**	15	84	

APPENDIX 4

Table A4.1 Hemeroby code, class and descriptions for land and seafloor
Adapted from Brentrup et al. (2002) and Fehrenbach et al. (2015).

Code	Class	Description and indicative example of land use	Description and indicative example of seafloor use
0	Natural	Undisturbed ecosystem, pristine forest, no utilisation	- Seafloor in pristine or near pristine condition subject to only minor indirect influence, e.g. marine debris. Habitats types include highly remote (e.g. oceanic atolls) and very deep habitats (lower continental slope, continental rise and abyss - depths below 2000m). - No fishing influence
1	Close-to-nature	Hardly influenced primary forests and their natural succession levels	- Seafloor in a natural state and populated by natural species; negligible historical direct impact. - Minor (localised or short-term) fishing influence, e.g. anchoring, single line methods.
2	Partially close-to-nature	Intermediate forest management (moderate thinnings, natural assemblage of species); Highly diversified agroforestry systems, low input	- Seafloor used routinely for human uses - E.g. low to moderate intensity activities e.g. demersal trawl in resilient habitat, Danish seine in unconsolidated sediments.
3	Semi-natural	Semi-natural forest management (regular thinning, exotic species); medium intensity extensive grassland, orchards, highly structured cropland with low input	- Intensively used habitats, e.g. historical fishing grounds for bottom contact fishing methods in moderately resilient habitat - E.g. moderate to high intensity demersal trawling
4	Partially distant-to-nature	Mono-cultural forest; intensive agricultural land use, short rotation coppices	- Original habitat largely removed, destroyed or permanently altered, especially where there are vulnerable and slowly recovering biota such as large and erect fauna including corals and sponges, and in areas of low productivity including deep continental slopes (depths >200 m); natural biota severely impacted or replaced by invasive or exotic species - E.g. destructive practices: dynamite and cyanide fishing, high-intensity demersal trawl, scallop dredging
5	Distant-to-nature	Distant-to-nature agricultural land use, landfill and dump sites, partly built-up areas, strong and long-term modification of biotopes	- No resemblance to original habitat e.g. dredged for sand or highly polluted, with no original biota or communities, - Fishing influence prevents regeneration
6	Non-natural artificial	Long-term sealed, degraded or devastated area (i.e. no habitat for plants)	- Reclaimed land with no habitat for marine species, permanent hypoxic ‘dead’ zones - No relevant fishing influence

Table A4.2 Hemeroby code, class and descriptions for land and seawater column, adapted from Brentrup et al. (2002) and Fehrenbach et al. (2015).

Code	Class	Description and indicative example of land use	Description and indicative example of seawater column use
0	Natural	Undisturbed ecosystem, pristine forest, no utilisation	- Seawater column in pristine or near pristine condition - no fishing influence
1	Close-to-nature	Hardly influenced primary forests and their natural succession levels	- Seawater column in natural state, natural species composition - limited removal of species through very low intensity fishing
2	Partially close-to-nature	Intermediate forest management (moderate thinnings, natural assemblage of species); Highly diversified agroforestry systems, low input	- Seawater column routinely used for fishing - E.g. low to moderate intensity purse-seine
3	Semi-natural	Semi-natural forest management (regular thinning, exotic species); medium intensity extensive grassland, orchards, highly structured cropland with low input	- Intensively used seawater column - E.g. moderate to high intensity mid-water-trawl
4	Partially distant-to-nature	Mono-cultural forest; intensive agricultural land use, short rotation coppices	- Permanently altered seawater column, natural ecosystem severely impacted, especially where there are vulnerable and slowly recovering species, or replaced by invasive or exotic species - E.g. destructive practices or overfishing
5	Distant-to-nature	Distant-to-nature agricultural land use, landfill and dump sites, partly built-up areas, strong and long-term modification of biotopes	- Seawater column ecosystem highly modified, no resemblance to original habitat e.g. highly polluted, with no natural biota or communities - Fishing influence prevents regeneration
6	Non-natural artificial	Long-term sealed, degraded or devastated area (i.e. no habitat for plants)	- No remaining ecosystem structure or function, e.g. Reclaimed land with no habitat for marine species - No relevant fishing influence

A4.1 Clarification of terms used

Table A4.3. Guidance for confidence terms - adapted from MSC (2010)

	Types of evidence
Low degree of confidence	Plausible argument, across a range of viewpoints and hypotheses. Based on analogy from similar situations with limited direct observations from the fishery (e.g. qualitative or general observations). Substantially relies on qualitative assessment and expert judgement.
High degree of confidence	Plausible argument and interpretation of direct observations across a range of viewpoints and hypotheses. Based on analogy from similar situations that is supported by significant direct observations from the fishery. Relies on an even balance of qualitative assessment/expert judgement and quantitative assessment.
Evidence	Quantitative inclusion of uncertainty and reasonable alternative hypotheses. Based mainly on direct observations from the fishery, with limited reliance on analogy. Substantially relies on quantitative assessment.

Definitions of terms

Serious harm: relates to gross change in habitat types or abundances, and disruption of the role of the habitats (MSC 2010).

Unlikely to be reversible: changes are expected to take much longer to recover than the dynamics in unfished situations would imply (e.g. some sort of regime change is implied from which recovery may not automatically occur). Examples of serious or irreversible harm include the loss (extinction) of habitat types, depletion of key habitat forming species or associated species to the extent that they meet criteria for high risk of extinction, and significant alteration of habitat cover/mosaic that causes major change in the structure or diversity of the associated species assemblages (MSC 2010).

Likely to cause serious harm: it assumed that with current fishing practices (gear, intensity) that serious harm will occur in the ecosystem or habitat, although there is no confidence around this assumption.

Unlikely to cause serious harm: it assumed that with current fishing practices (gear, intensity) that serious harm will not occur in the ecosystem or habitat, although there is no confidence around this assumption. Depending on the substrate and benthic communities affected, the harm may be reversible in the short-term, long-term or be unlikely to recover. Confidence around the type of substrate and benthic communities can help discern between hemeroby classes. For e.g. trawling on sandy substrate prone to natural disturbance in a data rich fishery may result in the score 'Fishery unlikely to cause serious harm, effects are reversible in the short-term, high degree confidence'. While a similar fishery which is data poor may be scored 'Fishery unlikely to cause serious harm, effects are reversible in the short-term, low degree confidence'.

Unlikely species could be seriously depleted: it assumed that with current fishing practices (gear, intensity) that serious depletion of retained or bycatch species will not occur, although there is no confidence around this assumption.

Level of harm unlikely to impact protection and rebuilding: it assumed that with current fishing practices (gear, intensity) that protection and rebuilding of endangered, threatened and protected species will not be impacted, although there is no confidence around this assumption.

APPENDIX 5

Table A5.1 Studies with assessments of actual or modelled diets

Author/year/ country	Diets modelled	Environmental impacts assessed	Seafood species assessed	Data sources (impacts of seafood)	Key results & conclusion	Discussion of seafood sustainability
(Donati et al. 2016) Italy	Current diet compared with low cost, environmentally sustainable diet and sustainable diet scenarios	GHGe, water, land (amount to regenerate the resource)	Not stated	Barilla Centre	-An optimisation tool was used to identify sustainable diet -Model suggests complete substitution of meat and fish with vegetal proteins	No
(Gephart et al. 2016) USA	Use footprints of food products to calculate minimised diets	GHGe, nitrogen release, water use (blue and green water), and land use	Not stated	Feed-use data from (Tacon et al. 2011) GHGe data based on (Heller and Keoleian 2015) (from Dutch LCA food database)	-Plant-based food and seafood (fish and other aquatic foods) commonly appear in minimised diets -emphasis on seafood is complicated by the environmental impacts of aquaculture versus capture fisheries	Increased seafood consumption suggested by the results would likely be met with increases in the footprints for seafood due to water, nitrogen and land footprints of aquaculture
(Horgan et al. 2016)	Two diets, one to meet dietary guidelines and the other to reduce GHGe, based on minimised current intake of individuals	GHGe	White and oily fish	(Audsley et al. 2009)	- The healthy diets and sustainable diets produced a 15 and 27% reduction in GHGe respectively - fish was one of the most commonly added foods to make an individuals’ current intake sustainable	The dietary recommendation to increase consumption of fish, for example, does not take into account the conflict between demand and the sustainability of fish stocks
(Tom et al. 2016) USA	Current consumption compared with reducing calories, USDA dietary recommendations without reducing calories, reduce calories and USDA	GHGe, energy, water	Range of wild-capture and aquaculture	Meta-analysis of LCA literature	-Diets with reduced Caloric intake and a shift to the USDA recommended food mix increase impacts -due to USDA recommendations for greater Caloric intake of fruits, vegetables, dairy, and fish/seafood, which have relatively high resource	No

	dietary recommendations				use and emissions per Calorie	
(Röös et al. 2015) Sweden	Current average Swedish diet, diet corresponding to Nordic recommendations, lifestyle Low Carbohydrate-High Fat (LCHF) diet	GHGe, land use, biodiversity damage potential	Not stated	(Röös 2012) in Swedish	Comparisons based on nutrient density scores depended on the score used, but the current and LCHF diets had more impact than the recommended diet (less livestock products) for all but one score	No
(Green et al. 2015) United Kingdom	Current UK diet, WHO recs, diet with fewer animal products and processed snacks and more fruit, vegetables and cereals	GHGe	Not stated	(Iribarren et al. 2011)	GHG emissions would be reduced if average diets among UK adults conformed to WHO recommendations, Further GHG emission reductions of around 40 % could be achieved by making realistic modifications to diets to contain fewer animal products and processed snacks and more fruit, vegetables and cereals	No
(Hess et al. 2015) United Kingdom	Current UK diet, 5 alternative healthier diets	Water (blue and green virtual water)	Not stated	None (VW was assumed to be zero for fish; fish combined with meat, eggs, beans and other non-dairy sources of protein)	If current trade patterns continue, policies to promote healthier eating in the UK may contribute to increased blue water scarcity at home and in other parts of the world	No
(Ruini et al. 2015) Italy	Omnivorous, lacto-ovo-vegetarian, vegan	GHGe, water (virtual water content), Ecological footprint	Not stated	Barilla Center	A diet based on the principles of the Mediterranean diet generates a lower environmental impact compared to diets that are heavily based on daily meat consumption	sustainability of fishing remains a concern
(Temme et al. 2015) Netherlands	Current diet, high and low environmental load diets	GHGe	Not stated	Blonk Consultants data set, version 2012	Higher GHGE of daily diets was associated with higher intakes of plant-based foods and even higher intakes of animal foods	No
(Tyszler et al. 2015)	Current diet of an average woman	GHGe, energy, land use	Herring, farmed salmon, cod,	(Kramer et al. 2013)	-Removing meat and fish from the diet reduces the environmental	The involved species, such as herring and farmed salmon, are

Netherlands	age 31–50 compared with six diet scenarios		mackerel, Pangasius, shrimp, tuna, whitefish (unclear if GHGe, energy and land use calculated for each species)		impact by about 21 % -Solutions without fish are impossible if model must meet dietary guidelines for omega-3 -Eating of legumes and (sustainable) fatty fish should be promoted -lack of a good indicator for marine resource depletion, and ignoring this indicator creates a bias towards higher consumption of fish	currently not overfished
(van Dooren and Aiking 2015) Netherlands	Present Dutch diet, Mediterranean, and New Nordic Diet	GHGe, land use	Not stated	Not stated (for seafood)	-An optimised Low Lands Diet has the same healthy nutritional characteristics (Health Score 123) as the Mediterranean Diet (122) and results in a lower environmental impact than the Mediterranean and New Nordic Diet -The consumption of fish and white meat should be increased and the intake of cheese and beef has to be reduced.	No
(Masset et al. 2014b) France	Lower-carbon, higher-quality and more sustainable diets defined	GHGe	Not stated	Greenext Service consultants assigned the GHGE values for 391 foods	Lower-carbon and more sustainable diets had reduced animal products, including fish	No
(Saxe 2014) Denmark	New Nordic Diet (NND) and Average Danish Diet (ADD)	16 impact categories assessed, data presented on respiratory inorganics, land use (nature occupation), and GHGe	Not stated	Danish LCA-Food database and the Ecoinvent database version 2.2	-Reducing the content of meat and excluding most long-distance imports were of substantial environmental and socioeconomic advantage to the NND when compared with the ADD -the recommendation to double the consumption of fish and seafood negatively affects the environment (increased emission of respiratory inorganics and greenhouse gases)	No
(Scarborough et al. 2014) United Kingdom	self-selected meat-eaters, fish-eaters, vegetarians and	GHGe	Range of seafood names with same	(Audsley et al. 2009)	-Dietary GHGe in self-selected meat-eaters are approximately twice as high as those in vegans	No

	vegans		GHGe value		-Fisheaters have lower GHGe than meat eaters and higher than vegetarians	
(van Dooren et al. 2014) Netherlands	Current average Dutch, official 'recommended' Dutch, semi-vegetarian, vegetarian, vegan and Mediterranean diets	GHG and Land use			-The Mediterranean diet is generally the health focus option with a high sustainability score -semi- and pesco-vegetarian are the options with the optimal synergy between health and sustainability	Trade-off between health and sustainability due to the role of fish and dairy in the diet
(Hendrie et al. 2014) Australia	Average Australian, Average Australian with minimal non-core foods, and diet based on Australian dietary guidelines	GHGe	Not stated	Australian national input-ouput tables and national greenhouse gas inventory, using AUS-MRIO model	-Diet based on Australian dietary guidelines had lowest GHGe -Red meat and non-core foods made greates contribution to diet related GHGe -Fish made small contribution to GHGe in all diets	No
(Jalava et al. 2014) global	Current diet with 4 scenarios based on dietary guidelines	water	Wild-capture	No data used	-Reducing animal products in the human diet offers the potential to save water resources, -fish retained in no animal diet	As many of the world's wild fisheries are already overexploited fish consumption was constrained to its current level and not increased over time
(Tilman and Clark 2014) global	Omnivorous, Mediterranean, Pescatarian (no meat), Veg	GHGe, Land use	Broad range of seafood	20 published seafood LCAs	Pescatarian diets have lower GHGe than Mediterranean and omnivorous diets, as well as require less land than the 2050 global-average per capita income-dependent diet -Global adoption of the Mediterranean or the pescatarian diet by 2050 would require 62% or 188% more seafood production, respectively. If wild-caught landings stayed at current levels, aquaculture would have to increase at 4.1% per year from 2010 to 2050 to meet the demand of the pescatarian diet.	No
(Saez-Almendros	Spanish current diet,	GHGe, land use, energy	Not stated	LCA food	Mediterranean diet has a reduced	No

et al. 2013) Spain	Mediterranean diet, Western diet	consumption and water consumption		database, Spanish Government, Heller 2000	footprint compared with regular diet, Regarding GHG emissions; fish also showed a remarkable environmental contribution in all the dietary patterns, third main contributor to GHG emissions in all dietary patterns after meat and dairy	
(Wilson et al. 2013) New Zealand	Modelled 16 diets	GHGe	Not stated	UK data (Berners-Lee et al. 2012)	-Identified optimal foods and dietary patterns that would lower the risk of non-communicable diseases at low cost and with low GHGe -found optimised diets that excluded meat and fish	No
(Vieux et al. 2013) France	Modelled diets of 1918 people	GHGe	Not stated	Greenext consultancy assigned values for GHGEs	-High-nutritional-quality diets had significantly higher GHGEs than did low-nutritional-quality diets. Despite containing large amounts of plant-based foods, -highest GHGe value recorded for the ruminant meat food group followed by fish food group - results confirm that animal-based products (ruminant meat, fish, dairy products, and pork, poultry, and eggs) have higher GHGEs than do plant-based products (fruit and vegetables and starchy food) on a weight basis (8)	No
(Macdiarmid et al. 2012) United Kingdom	Modelled a sustainable diet	GHGe	Fish and shellfish given same GHGe values	(Audsley et al. 2009)	A sustainable diet that meets dietary requirements for health with lower GHGEs can be achieved without eliminating meat or dairy products or increasing the cost to the consumer.	One of the most controversial areas in balancing health and environmental concerns is in relation to fish consumption
(Tukker et al. 2011)	European status quo and three simulated diet baskets: universal dietary recommendations, the same pattern with reduced meat	Abiotic resource depletion, GHGe, Ozone depletion, Human toxicity, Ecotoxicity, Photochemical oxidant Formation,	Not stated	E3IOT environmentally extended input output database	The positive effect of limited reductions in meat intake appears to be cancelled out by enhanced intake of fish, cereals, and vegetables.	It has to be noted that E3IOT is not capable of assessing the impacts on biotic depletion. Negative impacts of enhanced fish consumption in scenario 3 are hence not fully taken into account.

	consumption, and a 'Mediterranean' pattern with reduced meat consumption	Terrestrial acidification, Freshwater Eutrophication				
(Fazeni and Steinmüller 2011) Austria	Baseline and 1 sustainable scenario	Land use, energy demand, and GHGe	Not stated	Global Emission Model for Integrated Systems [GEMIS]	Compliance with healthy eating guidelines leads to lower energy demand and a decrease in GHGe, largely due to a decrease in livestock numbers	It is assumed that there is no potential, in view of depleted fish stocks, to increase the supply of fish from the world's oceans. The lack of omega-3 and omega-6 fatty acids is made good with vegetable oils.
(Stehfest et al. 2009)	"business as usual" reference diet, 4 variants: complete substitution of meat from ruminants (NoRM), complete substitution of all meat (NoM), complete substitution of all animal products (meat, dairy products and eggs) (NoAP) and partial substitution of meat based on a healthy diet variant (HealthyDiet, HDiet).	Land use, GHGe	Not stated	Not stated	Global food transition to less meat, or even a complete switch to plant-based protein food to have a dramatic effect on land use -Healthy diet variant - consumption of fish, poultry and eggs is advised with zero to two servings per day.	Model does not allowing global total fish consumption to increase
(Eshel and Martin 2006) USA	Several semirealistic mixed diets: mean American, red meat, fish, poultry, and lacto-ovo vegetarian	GHGe	Herring, tuna, salmon, shrimp	GHGe values based on energy efficiency (as the percentage of fossil fuel input energy that is retrieved as edible energy)	-The fish diet results in lower GHG emissions than both the red meat and mean American diets – -noted equality of fish and red meat efficiencies which reflects the large energy demands of the long-distance voyages required for fishing large predatory fishes such as swordfish and tuna toward which western diets are skewed, and the relatively low energetic efficiency of salmon farming	No

Table A5.2 Studies assessing products as part of sustainable diets

Author /year/country	products modelled	Environmental impacts assessed	Seafood species assessed	Data sources (impacts of seafood)	Key results & conclusion	Discussion on seafood sustainability
(Nijdam et al. 2012) global	Beef (intensive , extensive, from dairy cows), pig, poultry, sheep meat, milk, eggs, seafood from fisheries and aquaculture, vegetal protein, other meat substitutes	GHGe, land use	Fisheries and aquaculture	16 life cycle assessment studies of fisheries and aquaculture	-The range of the carbon footprint is especially large for beef products and seafood -feed production and animal husbandry are the most important contributors to the environmental impacts	Fishing can have large impacts on marine ecosystems. The removal of large amounts of fish directly affects predators, predated species and competitors. Bottom trawling generally has very high discard rates and destructive effects on the seabed
(González et al. 2011) Sweden	84 common food items of animal and vegetable origin	Energy use, GHGe	Tuna and salmon	(Pelletier et al. 2009; Hospido and Tyedmers 2005)	-Animal-based foods are associated with higher energy use and GHGe than plant-based foods, with the exception of vegetables produced in heated greenhouses -GHGe for fish were lower than beef, mutton, pork and chicken per kg but higher per gram of protein	No
(Masset et al. 2014a)	363 most commonly consumed foods in France	GHGe, acidification, and eutrophication	Not stated	Standard lifecycle inventory data e.g. Ecoinvent	- Meat, fish, eggs and dairy products had the strongest influence on the environment -Fish products had the most variable GHGe but achieving much higher nutritional quality values than meat and poultry. -identifying the most sustainable foods within each food group may be a sensible option	No
(Drewnowski et al. 2015)	483 foods and beverages	GHGe	Not stated	GHGe values were calculated and provided by a French supermarket chain	-On a per-weight basis (100 g), processed meats, meat dishes, cheeses, and processed fish were associated with higher GHGe values -The more–nutrient-dense animal products, including meat and dairy, had higher GHGE values per 100 g but much lower values per 100 kcal	No
(Carlsson - Kanyama	study of 20 items sold in Sweden	GHGe	Herring, cod	carbon dioxide emissions were calculated	-Changes in the diet toward more plant-based foods, toward meat from animals with little enteric fermentation, and toward foods	Cod high GHGe because overfished and extensive use of

and González 2009)			based on an energy analysis	processed in an energy-efficient manner offer an area for mitigating climate change, -Fish may or may not present high emissions of carbon dioxide due to fossil fuel use - certain fish are found in the midrange GHGe while some in the low range e.g. herring	fuel for trawling, cod fishing is still profitable because of heavy subsidies for fisheries in the European Union
(Weber and Matthews 2008)	GHGe	Not stated	emissions/impact data for the calculation is in terms of emissions per industrial output, as is standard in IO-LCA	-Red meat is around 150% more GHG intensive than chicken or fish. -Shifting less than one day per weeks' worth of calories from red meat and dairy products to chicken, fish, eggs, or a vegetable-based diet achieves more GHG reduction than buying all locally sourced food	No

Table A5.3 reviews of sustainable diet literature

Author/year/ country	Studies reviewed	Key findings	Discussion on seafood sustainability
(Jones et al. 2016)	A Systematic Review of the Conceptualisation and Measurement of Sustainable Diets	-Estimating the GHGEs of foods using LCA was the most common method used to measure the environmental impacts of diets -Many components of sustainable diets identified in existing conceptual frameworks are disproportionately underrepresented	Fisheries listed as component of sustainable diets, with focus on overfishing, aquaculture management and aquaculture feeding
(Reynolds et al. 2014)	Environmental impact assessment and LCA literature on environmental impacts of dietary recommendations	-Most studies support environmental benefits of a reduced consumption of animal-based foods and increased consumption of fruit and vegetables -In general, adhering to dietary guidelines reduces impacts on the environment	The case can easily be made that reducing the intake of animal protein (including from fish) and dairy foods in the global diet would potentially have considerable impact on reducing environmental effects.
(Heller et al. 2013)	32 studies that use LCA framework to evaluate environmental impacts of diets or meals	-Need expansion of scope of assessments beyond the current focus on GHGe. -Typical food industry sectors are at the level of "red meat," "chicken, fish, and eggs," "dairy products," "fruits and vegetables," and "cereals and carbs" and therefore typically do not allow detailed exploration of dietary choices	Can fish stocks support recommended consumption levels? Further examination of the role of sustainable aquaculture in meeting fish demand needed

Table A5.4 Discussion papers that address seafood sustainability and sustainable diets

Author/year	Paper focus	Conclusions	Comments on seafood sustainability
(Merrigan et al. 2015)	Incorporation of food system sustainability in official dietary guidance	-Need to incorporate sustainability into dietary guidelines -future food insecurity is predictable without attention to sustainability -dietary advice for health and sustainability is the same: eat less meat -Challenge is how to produce the most healthful foods in a way that sustains employment in the agricultural sector and minimises adverse impacts on the environment	Refers to Dutch 2011 dietary guidance advice to eat two portions of fish per week which was deemed "ecologically Detrimental' and Dutch Health Council is now evaluating the sustainability of individual fish species for new version of guidelines.
(Alsaffar 2015)	Explores interaction between food industry, nutrition, health and the environment.	-A healthy and sustainable diet would minimise the consumption of energy-dense and highly processed and packaged foods, - urgent need to develop and promote strategies for sustainable diets - include less animal-derived foods and more plant-based foods and encourage people not to exceed the recommended daily energy intake	Increasing fish consumption is an example of an ethical dilemma; it would improve health but have a negative effect on fish stocks.
(Lang 2014)	Reviews the case for sustainable diets	-Coherence of data, principles and purpose is needed at the global and regional policy-making levels to reduce the system's negative impact on health, environment and economies -counterbalances current dominant policy emphasis on raising food output as the best route to a sustainable food future.	Nutrition guidelines worldwide encourage the consumption of fish and fish oil however over half global wild fish stocks "fully exploited"
(Macdiarmid 2013)	Examines links between healthy and environmentally sustainable diets	-Possible to achieve a realistic diet that meets dietary requirement for health and has lower GHGe, it cannot be assumed that a healthy diet will always have lower GHGe -understanding of sustainable diets is poor, which could contribute to the barriers towards changing dietary intakes	-Dietary recommendation for fish intake is the most widely recognised conflict between health and environmental sustainability -confusion among consumers about the sustainability of eating fish, despite the introduction of labelling of fish from sustainable sources and media campaigns - -good example of where a single consistent message about health and the sustainability of foods is needed otherwise the consumers will simply disengage
(Buttriss and	Considers need	UK Foresight report identifies a number of priorities	-Major concerns about the sustainability of fish

Riley 2013)	to embed thinking about nutrition into discussions about sustainability of food supply	for policy makers: as: (1) balancing future demand and supply sustainability; (2) ensuring there is adequate stability in food supplies; (3) achieving global access to food and ending hunger; (4) managing the food system so as to mitigate the impact of climate change; (5) maintaining biodiversity and ecosystems while feeding the world	supplies as a number of the world's fisheries are currently depleted, -emphasis is on increasing the diversity of species that are caught and placed on sale -ensure long term sustainability of fish stocks
(Selvey and Carey 2013)	Environmental impact of Australia's dietary guidelines	- Continuing to consume food that has a large ecological footprint will threaten our future food supply -guidelines need to do more for the health of the planet and the population	-Clear evidence for need to make stronger statements about eating fish: stocks globally are collapsing at an alarming rate, with more than three-quarters overexploited or overfished. - Aquaculture has expanded and largely relies on fishmeal, further depleting fish stocks - Climate change and ocean acidification will also have an impact on fisheries - fish consumption in Australia would need to increase by 40% To meet recommended intakes - Australian guidelines should be modified to match population consumption levels that are achievable within catch limits from Australia's exclusive economic zone, and should suggest alternative sources of omega-3 fatty acids
(Clonan and Holdsworth 2012)	Explores concept of incorporating the sustainability aspects of the human diet with those of nutrition	-Changes are necessary to existing food-based dietary guidelines to reflect reductions necessary within certain food groups—for example, meat and dairy foods—and increases within others—for example, bread, rice, and potatoes -specialists should consider broadening criteria to include the food consumption process to build on the link between healthier dietary intakes and attitudes toward sustainable food	-The future sustainability of current protein sources such as meat and fish remains one of the biggest challenges for a sustainable food system -this qualification has been reflected internationally in the nutrition policy agendas of some European countries e.g. German and Swedish Guidelines both recommended eating less meat and fish
(Clonan et al. 2012)	Investigates whether health and/or sustainability are motivating factors when purchasing and consuming fish	The number of consumers purchasing fish for health reasons was more than those seeking sustainably sourced fish; yet, they still failed to meet the recommended intake set by the Food Standards Agency -Clear advice should be communicated enabling consumers to meet nutritional needs while protecting fish stocks.	-Fish stocks under pressure, three-quarters fully or over-exploited -Dietary advice to the public to increase consumption of fish conflicts with the prevailing pressure on fish stocks. -report of the Council of Food Policy Advisors highlights fish consumption as a core issue and recommends shifting targets for consumption towards produce that has come from

			only sustainably managed stocks, eliminating the consumption of threatened species
(Riley and Buttriss 2011)	Examines links between healthy and sustainable diets	-Challenge to identify dietary patterns that provide many nutrients in appropriate amounts, that are also equitable, affordable and sustainable -no consensus on the details of how people's diets should change from a sustainability perspective -important that public health nutritionists continue to voice the need for both sustainability and health factors to be considered together	-Guideline is in potential conflict with concerns over the sustainability of global fish stocks following years of overfishing of some species in some locations - guidelines suggestion that people who regularly eat a lot of fish should try to choose as wide a variety as possible people are encouraged to experiment with less familiar species for which stocks are believed to be more abundant, such as coley, gurnard and mackerel
(Garnett 2011)	Examines opportunities for reducing greenhouse gas emissions in the food system	-Need to shift away from away from diets rich in GHG-intensive meat and dairy foods - priority for decision makers is to develop policies that explicitly seek to integrate agricultural, environmental and nutritional objectives	-Focus is on GHG emissions and a broader definition of 'sustainable consumption' will need to cover fish sourcing -risk of increase pressure on stocks if meat consumption lowered
(Mitchell 2011)	Explores dilemma of eating seafood	- Potential juxtaposition between the health benefits of fish consumption with concerns over resource capacity and sustainability	Providing we follow a few simple guiding principles, increasing our consumption of seafood in the cause of good nutrition and in the interests of national health is not necessarily contrary to good environmental stewardship of the oceans
(Westhoek et al. 2011)	Examines consumption and production of meat, dairy and fish in the European Union	- Meat, dairy, eggs and fish provide essential nutrients and have large environmental effects - Conversion of plant energy and proteins into edible animal products is a generally inefficient use of resource - options to reduce the impacts of livestock production are: shifts in consumption, resource efficiency and producing with fewer local impacts	- Many marine fish populations are overexploited. despite new fishing grounds, EU catches are declining rapidly -switch to an increased consumption of herbivorous fish, would reduce the amounts of wild-caught fish required in fish feed

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Farmery, A., Gardner, C., Green, B. S., Jennings, S., Watson, R. W., 2015. Life cycle assessment of wild capture prawns: expanding sustainability considerations in the Australian northern prawn fishery, Journal of cleaner production, 87, 96-104

Farmery, A., Gardner, C., Green, B. S., Jennings, S., 2014. Managing fisheries for environmental performance: the effects of marine resource decision-making on the footprint of seafood, Journal of cleaner production, 64, 368-376

Farmery, A. K., Gardner, C., Green, B. S., Jennings, S., Watson, R. A., 2015. Domestic or imported? : An assessment of carbon footprints and sustainability of seafood consumed in Australia, Environmental science & policy, 54, 35-43

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APPENDIX 8 ALL PUBLICATIONS PREPARED AND PUBLISHED DURING CANDIDATURE

Farmery, A, Gardner, C, Green, BS, Jennings, S & Watson, R 2015 'Domestic or imported? An assessment of carbon footprints and sustainability of seafood consumed in Australia', *Environmental Science & Policy,* vol 54, pp. 35-43.

Farmery, A, Green, BS, Jennings, S, Watson, R & Gardner, C 2015 'Life cycle assessment of wild capture prawns: expanding sustainability considerations in the Australian Northern Prawn Fishery' *Journal of Cleaner Production*, vol. 87, pp. 96-104.

Farmery, A, Gardner, C, Green, BS & Jennings, S 2014, 'Managing fisheries for environmental performance: the effects of marine resource decision-making on the footprint of seafood', *Journal of Cleaner Production*, vol. 64, pp. 368-376.

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